The impact of maize silage production and supplementary feed use on the carbon balance of New Zealand dairy farms

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Abstract

Globally, agriculture contributes 10-12% to anthropogenic greenhouse gas (GHG) emissions. Consequently, mitigation of agricultural GHGs has taken on increased importance, particularly in countries like New Zealand where agriculture accounts for almost half of national emissions. Sequestration of atmospheric CO_2 in the soil by altering land management practices has been identified as a potential mitigation option for anthropogenic GHG emissions. However, implementing management practices as a mitigation option first requires an understanding of their effect on soil C stocks. Often the effects of cropping and grassland management on soil C stocks are studied individually, but these practices are linked when the resulting crop is supplied to grazing animals as supplemental feed. This is important for the New Zealand dairy industry, which has been traditionally pasture-based, but increasingly is supported by supplemental feed to compensate for periods of low pasture growth and to boost productivity. To understand the impact of supplemental feed use on soil C stocks and, therefore, on any mitigation potential, both the production of the feed and its use need to be considered together. The overarching aim of this thesis was to experimentally determine the impacts of supplemental feed production and use on soil C by using the net ecosystem carbon balance (NECB) methodology to quantify changes in ecosystem carbon (C) stocks (assumed synonymous to the change in soil C). A secondary aim of this thesis was to advance NECB methodology in complex grazed pasture systems, primarily through examination of the scale at which measurements were made. Improved methodology and understanding are needed to allow a greater number of management practices to be tested.

Conceptually, importation of supplemental feed and its embodied C can lead to an increase in ecosystem C because consumption of supplemental feed C by the animals results in additional excreta deposition on the pasture during grazing that can be stored as soil C. This hypothesis was tested by determining the NECB for three years on a dairy farm where imported supplemental feed accounted for >40% of the cows' diet. A positive NECB (indicating a gain of ecosystem C) was calculated for all three years, but consideration of uncertainties resulted in only one year having a definitive gain of C. The three-year average NECB was 71 ± 77 g C m^{-2} y^{-1} (mean ± uncertainty) and was not considered different from zero. Theoretical calculations based on the imported quantity of supplemental feed C (average 526 g C m^{-2} y^{-1}) coupled with the digestibility of the feed and manure retention rates suggest gains of around 25 g C m^{-2} y^{-1} could be expected. The results of this study were of the same order

of magnitude to what was expected from modelling and manure C retention literature, and although experimentally a gain in C associated with a large import of supplemental feed could not be definitively concluded, the results confirmed that large gains of ecosystem C are unlikely.

A broad range of supplemental feed is used within the New Zealand dairy industry including grazed and harvested feeds, with maize harvested for silage being one of the more common. Internationally, sites where maize cropping with full biomass harvest occurs have been identified as a large source of C, but these studies tend to be from long-term cropping systems. Within New Zealand dairy farm systems, maize silage is often grown as part of the pasture renewal process and, consequently, findings from studies within long-term cropping systems may not apply. In this study, NECBs were calculated for a system where a long-term pasture site was converted to maize silage cropping for two years before a return to permanent pasture. To isolate the C balance of the maize crop alone, NECBs were calculated for the period of maize crop establishment through to seedling emergence of the subsequent sward (~190 days). The Year 1 maize crop NECB was –850 g C m^{-2} (a loss of C), while the Year 2 maize crop lost a further –415 g C m^{-2}. Concurrent grazed pasture NECBs from the same farm were 11 g C m^{-2} and –115 g C m^{-2} over the same two periods. Above-ground biomass production was around three times greater from the maize crop than adjacent pastures, with more than 90% of this production exported from the site, compared to around 60% net export of the pasture biomass after accounting for returned excreta. The hypothesis that a large loss of ecosystem C could be expected from maize silage cropping for supplement feed was supported. Future research to determine whether the return to permanent pasture results in recovery of previously lost C are required to understand the long-term impacts of periodic cropping for supplemental feed production.

Consideration of the effect that all types of supplemental feed production have on ecosystem C stocks was beyond the scope of this thesis, but conclusions can be drawn on systems which use maize silage. Dairy farms which import supplemental feed (maize silage or other) are likely to see small increases (<50 g C m^{-2} y^{-1}) in the ecosystem C stocks regardless of the quantity imported, while the production site would be expected to have large losses when producing maize silage. Results from this thesis suggest that where production and use occur within the same dairy farm system a net loss would be expected, and if averaged across the entire farm would be in the order of –40 g C m^{-2} y^{-1}. Losses during maize silage production may be reduced by minimising the time that soil is bare during establishment, and although not tested,

possibly by decreasing tillage intensity. Moreover, if ecosystem C losses during production are recovered longer-term when returned to grazed pasture (i.e. in the several years following cropping), on-farm production of periodically cropped maize silage may lead to small, but consistent gains in soil C and provide the potential for GHG mitigation. A key unresolved question is the rate of C recovery following a return to permanent pasture relative to the cropping return period.

While determining the effect of supplemental feed on ecosystem C, the opportunity also arose to investigate two aspects of NECB measurement scale. Firstly, NECBs were compared when calculated with an ecosystem boundary equivalent to (i) the paddocks included within the eddy covariance (EC) flux footprint ($NECB_{Footprint}$), and (ii) the farm boundary ($NECB_{Farm}$). Both calculated NECBs were similar ($NECB_{Footprint}$ was 56 ± 77 g C m^{-2} y^{-1} and $NECB_{Farm}$ was 71 ± 77 g C m^{-2} y^{-1}) and the selection of the best boundary definition was dependent on the quality of the available data with $NECB_{Farm}$ considered best in this study. Furthermore, components contributing to the NECB differ with system boundary location and, therefore, can influence interpretation. When choosing a system boundary, the assumption that the measured CO_2 exchange is representative of the entire area within the defined boundary needs to be cautiously considered. The second methodology investigation calculated paddock-specific NECBs for two adjacent paddocks with a single EC system located between them. Provided regular EC data are available from both paddocks (i.e. due to regularly changing wind directions), paddock-specific NECBs can be calculated. Advantages of this approach include eliminating inherent management heterogeneity (e.g. asynchronous grazing), and the ability to allow for treatment comparisons or provide replication while minimising spatial variability and potentially reducing equipment requirements. Key disadvantages were a reduction in data coverage (from 49.1% for the full footprint to 25.9% and 15.7% for each paddock), an increase in uncertainty (by about 25%), and the need for prior assessment of site suitability (i.e. the need for regular wind from both paddocks). Comparisons of NECBs from adjacent rotationally grazed paddocks identified large inter-annual and between-paddock variability, with the latter often due to subtle management differences despite the same overall management regimes. Finally, due to the spatial and temporal variability, several measurement years would be needed to (i) determine the true trajectory of ecosystem C balances and (ii) determine similarity or differences between the two paddocks.

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Abbreviations

Abbreviation	Definition
ANN	Artificial neural network
C	Carbon
CH_4	Methane
CO_2	Carbon dioxide
d	Day
DM	Dry matter
EC	Eddy covariance
ER	Ecosystem respiration
ESM	Equivalent soil mass
g	Gram
GHG	Greenhouse gas
GPP	Gross primary production
h	Hour
ha	Hectare
HUE	Harvest use efficiency
H_2O	Water
K	Potassium
kg	Kilogram
km	Kilometre
m	Metre
mm	Millimetre
N	Nitrogen
NECB	Net ecosystem carbon balance
NBP	Net biome productivity
NEE	Net ecosystem exchange
NEP	Net ecosystem production
N_2O	Nitrous oxide
P	Phosphorus
PI	Phytomass Index
PKE	Palm kernel expeller
PPFD	Photosynthetic photon flux density
PWP	Permanent wilting point
R_N	Net radiation
s	Second
t	Ton
T_a /T_A	Air temperature
T_S	Soil temperature
VMC	Volumetric moisture content
VPD	Vapour pressure deficit
WFPS	Water-filled pore space
y	Year

Chapter 1:

Introduction

1.1 Introduction

Globally, agriculture occupies ~38% of land area, consisting of croplands (12%) and grasslands (26%) (e.g. FAO, 2019), and contributes an estimated 10 to 12% of total anthropogenic greenhouse gas (GHG) emissions (Smith *et al.*, 2014). GHG emissions from agricultural activities are predominantly in the form of nitrous oxide (N_2O) and methane (CH_4), with carbon dioxide (CO_2) emissions or uptake associated with land-use activity and change (Tubiello *et al.*, 2015). The effect of net CO_2 exchange with the atmosphere (gains or losses) is represented by a change in ecosystem carbon (C) stores, including biomass and soil organic matter.

Soils are the largest terrestrial store of C with ~2000 Pg C to 1 m depth, or about four times more than is stored in plant biomass (Janzen, 2004). Since the industrial revolution, there has been a significant loss of soil C to the atmosphere, with estimates suggesting 133 Pg C has been lost to 2 m depth (Sanderman *et al.*, 2018). Consequently, it is considered that the C stocks of soils have the potential to be increased by sequestering atmospheric CO_2 and thus contributing to the mitigation of global GHG emissions (Minasny *et al.*, 2017). Indeed, authors such as Soussana *et al.* (2010) have suggested soils have the potential to sequester significant atmospheric CO_2, leading to movements such as the "4 per mille Soils for Food Security and Climate" initiative (4 per 1000, 2020), which is estimated to have the technical potential to offset 20-35% of global anthropogenic GHG emissions (Minasny *et al.*, 2017). While there is continuing debate as to the size of potential gains (e.g. Baveye *et al.*, 2018; White *et al.*, 2018), the greatest sequestration potential of soils is from managed agricultural soils (Minasny *et al.*, 2017), many of which have been degraded from previous agricultural activities.

Understanding the consequences of different agricultural management activities on soil C stocks has been a focus of considerable research, with an increased shift toward identifying those practices leading to increased sequestration. Mitigation opportunities, and consequently prior research, differ between grazed grasslands and croplands due to each having differing management practices. For example, within grazed agricultural systems, studies considering the effect of management on soil C have included grazing intensity

(Klumpp *et al.*, 2011), rotational and continuous grazing (Oates and Jackson, 2014), fertilisation and manure application (Limin *et al.*, 2015), mechanical harvesting of pasture (Koncz *et al.*, 2017), irrigation (Hunt *et al.*, 2016) and pasture renewal (Rutledge *et al.*, 2014). Further studies have considered the mitigation potential of increased species diversity (Rutledge *et al.*, 2017a) and the introduction of earthworms (Schon *et al.*, 2015) amongst others (Whitehead *et al.*, 2018). Meanwhile, considerable focus within cropping systems has included investigations of (for example) tillage (Busari *et al.*, 2015), residue management (Blanco-Canqui and Lal, 2009), nutrient supply (Triberti *et al.*, 2008), cropland irrigation (Suyker and Verma, 2012) and use of cover crops (Poeplau and Don, 2015).

Use of supplemental feed within grazing systems (whether pasture-based or housed) results in a proportion of the intake excreted, and when returned to the land as either fresh excreta or collected, stored and redistributed as manure provides an opportunity for C to be stored in the soil. Indeed, Maillard and Angers (2014) suggest the retention of manure C is around 12%, while long-term trials of application of additional farmyard manure have identified increased soil C stocks (Johnston *et al.*, 2009; Johnston *et al.*, 2017). Consequently, increased excreta deposition due to increased animal feed intake through the use of supplemental feed could be expected to have a similar result. In a scenario modelling study of New Zealand dairy grazing systems, Kirschbaum *et al.* (2017) identified the increased use of imported supplemental feed to be a management practice with the potential to increase both soil C stocks and milk production within a dairy farming system. Increased use of supplemental feed has gradually occurred in the New Zealand dairy sector within the last few decades to increase productivity (DairyNZ, 2019). Traditional New Zealand dairy farming systems operated with year-round rotational grazing of pasture supplemented by stored or cropped feed at times of low pasture growth. Today, supplemental feed is used not just to account for seasonal shortfalls in the available pasture, but also year-round to support increased stocking rates and production of dairy products leading to improved profits (Doole, 2014). Supplemental feed production consists of that grown on-farm and imported from external sources (both within New Zealand and from other countries). The supplemental feed used can be characterised as two types: pasture supplements (most commonly silage and hay), and non-pasture supplements. Non-pasture supplements include animal feed that has been grown from crops e.g. maize (silage and green-feed), brassicas, grains, etc. In particular, the use of maize silage has increased substantially, and now makes up almost one-third of all pasture supplements (DairyNZ Economics Group, 2016). Production of supplemental feed has the potential to

negatively impact ecosystem C stocks through mechanisms including reduced excreta inputs (due to the absence of grazing at the site at which the feed is grown), absence of vegetative cover during establishment and harvesting phases (Rutledge *et al.*, 2017b), full biomass removal (Eichelmann *et al.*, 2016) and tillage (Conant *et al.*, 2007).

To evaluate the effect that increased use of supplemental feed has on soil C stocks and GHG mitigation potential requires consideration of both the production and use of the supplemental feed. Kirschbaum *et al.* (2017) focussed on how supplemental feed could contribute to increases in soil C stocks, but their model intentionally did not include the consequences of the production and subsequent export of the supplemental feed on soil C stocks from the site of production. Experimentally, the effect of using supplemental feed (including production and consumption) on soil C stocks in dairy systems has not been previously quantified and is a significant gap in the literature especially when considered a potential approach for increasing soil C stocks (Kirschbaum *et al.*, 2017). Quantifying the effect of the production of supplemental feed on soil C stocks is a broad question given the wide range of supplemental feed options available to farmers (both pasture and non-pasture). This thesis investigates two aspects of supplemental feed use: (1) the effect of large quantities of imported supplemental feed (including maize silage) on the ecosystem C balance of the farm receiving the imported feed; and (2) the effect of periodic maize silage cropping on the ecosystem C balance at the site of production including comparison with grazed pasture. While there have been several, mostly international, studies investigating the impact of maize cropping (including maize silage) on soil C, these are not transferable to New Zealand systems, especially where short-term periodic maize silage cropping as part of a pasture renovation programme occurs (Densley *et al.*, 2001). Furthermore, long-term continuous cropping systems vary in their management including (i) differing establishment (tillage) methods; (ii) variable residue management; and (iii) generally include winter management ranging from remaining fallow to winter cropping. Assessment of maize silage cropping on ecosystem C in New Zealand allows for improved insights into how production and transfer of supplemental feed can affect soil C stocks of New Zealand dairy systems.

1.2 Aims and objectives

The main aim of this thesis was to investigate how aspects of supplemental feed production and use within New Zealand dairy farming systems has changed soil C stocks and determine the role of supplemental feed in GHG mitigation. The focus was on the use of large quantities

of supplemental feed, and production of maize silage through on-farm periodic cropping (rather than long-term continuous cropping). Changes in soil C stocks were estimated using the net ecosystem carbon balance (NECB) method (Chapin *et al.*, 2006) by coupling measurements of CO_2 exchange determined using the eddy covariance (EC) technique (Baldocchi, 2003) to measurements and estimates of all other flows of C into and out of defined ecosystem boundaries. This technique calculates the change in ecosystem C, which is considered synonymous to the change in soil C by assuming the biomass C stock at the end of the measurement period is equivalent to that at the beginning (Paustian *et al.*, 2019).

While the overall aim was practically focused, the subject provided an opportunity to further develop methodology, understanding and interpretation of results derived from EC and NECB studies in rotationally grazed agricultural systems. A key assumption of EC based studies is the homogeneity of the ecosystem over which the measurements are made (Rannik *et al.*, 2012), and while grazed pastures may consist of homogenous vegetation, the management is often heterogeneous, especially when the measurement footprint includes multiple paddocks. Furthermore, a major constraint of EC studies is the lack of replication (Hill *et al.*, 2017) and/or the high resource cost of manipulative experiments (Eugster and Merbold, 2015) thus limiting the effectiveness of studies investigating mitigation options. Consequently, two aspects of measurement scale were also investigated within this thesis. Firstly, a comparison was made between NECB calculated for ecosystem boundaries equivalent to (i) paddocks included in the EC flux footprint, and (ii) the farm boundary. Secondly, the possibility of determining NECB for two adjacent paddocks based on data from a single EC system was examined to minimise the influence of management heterogeneity, while providing a method to increase replication and/or allow for paired treatment-control experiments.

Combining the experimental and methodological aspects of this thesis led to a series of research objectives as outlined below. Note that these objectives, and the subsequent thesis chapters, are ordered by subject matter rather than chronologically. The specific research objectives of this thesis were:

1. To quantify the NECB of a dairy farm importing large quantities of supplemental feed (>40% of the cows' diet). This objective tests the hypothesis that large imports of supplemental feed would result in a positive NECB (a presumed gain in soil C). Additionally, the choice of system boundary on the calculation of NECB and subsequent interpretation of the results is examined

by calculating the NECB for two system boundaries: (i) an area bound by the footprint of the eddy covariance system; and (ii) an area bound by the farm property boundary.

2. To calculate the NECB of periodic maize silage production for supplemental feed during the maize cropping period compared to land used for pasture grazing (using data obtained from Objective 3). The hypothesis for this objective is that the periodic cropping nature coupled with conventional tillage and high soil C stocks will result in a large loss of ecosystem C.
3. To determine the viability of calculating paddock specific CO_2 exchange and NECB of two adjacent paddocks using a single EC system, and thus providing a novel method to test the effect of management practices and/or increase replication.

1.3 Thesis structure

This thesis begins with the introduction (this chapter) followed by a literature review (Chapter 2), three research chapters addressing the specific objectives as outlined in Section 1.2 (Chapters 3-5) and concludes with a summary (Chapter 6). Chapters 3-5 are presented as the published manuscript (see below) and thus are independent of each other. Each chapter contains a brief literature review in the introduction, followed by methods, results, discussion and conclusion sections, and consequently, there is some repetition between chapters.

In the context of this thesis, Chapter 2 provides a review of existing knowledge of supplemental feed use and production (both in general terms and specifically for maize), including an overview of New Zealand dairy farming practices and history, along with a description of the NECB methodology. Chapter 3 presents the NECB of a dairy farm with large quantities of imported supplemental feed and examines the impact of system boundaries on NECB and its interpretation. Chapter 3 has been published by the journal *Agriculture, Ecosystems and Environment* (Wall *et al.*, 2019). Chapter 4 reports the NECB of maize silage production and contrasts this with the alternate land use of rotationally grazed pasture. Chapter 4 was published by the journal *Agriculture, Ecosystems and Environment* (Wall *et al.*, 2020a). Chapter 5 tests the viability of calculating paddock specific CO_2 exchange and NECB of two adjacent paddocks using a single EC system, while providing the comparative NECB from rotationally grazed pasture for comparison with maize silage production (Objective 2). This chapter has been published by the journal *Agricultural and Forest Meteorology* (Wall *et al.*, 2020b). Finally, Chapter 6 summarises the research findings of this thesis and identifies opportunities for future research.

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Chapter 2:

Literature Review

2.1 Introduction

Worldwide agriculture is estimated to account for 10 to 12% of total global anthropogenic greenhouse gas (GHG) emissions (Smith *et al.*, 2014), however, in New Zealand, agriculture is a significantly larger proportion of greenhouse gas emissions (48% of national emissions; MfE, 2020). Dairy farming in New Zealand utilises 2.6 million ha (22% of all agricultural land; Statistics New Zealand, 2018a) and typically occurs on the flattest and most productive lands of which, for some soil orders, have shown decreases in soil C over the past 20-30 years (Schipper *et al.*, 2014). Many New Zealand soils on which dairy farming occurs have been suggested to have the capacity for sequestration (Beare *et al.*, 2014; McNally *et al.*, 2017). Consequently, identifying mechanisms which have the potential to increase soil carbon (C) in dairy pastures can help mitigate greenhouse gas emissions (e.g. Soussana *et al.*, 2010) and also improve soil health (Lal, 2016). Recent New Zealand soil C research has investigated the potential of introducing earthworms (Schon *et al.*, 2015), planting diverse pastures (McNally *et al.*, 2015; Rutledge *et al.*, 2017a) and deep ploughing (Calvelo Pereira *et al.*, 2017) as mitigation strategies, while also considering the effects of irrigation (Hunt *et al.*, 2016; Moinet *et al.*, 2017) and pasture renewal (Rutledge *et al.*, 2014; Rutledge *et al.*, 2017b). These are all either current practices or future options that can be considered by the dairy industry. Scenario modelling by Kirschbaum *et al.* (2017) also suggested that soil C could be gained by increasing the amount of supplemental feed being used either through on-farm production or imported from external suppliers. The impact of supplemental feed use on soil C stocks is a research area that has yet to be investigated but requires the consideration of both the use and supply of the supplemental feed. While increases to the soil C stocks through increased importation of supplemental feed may occur, there may be offsetting losses of soil C at the location of production, and the balance of these management practices is the focus of this thesis and literature review.

2.1.1 Purpose and structure of the literature review

While there is frequently a desire to determine whether a specific management practice alters soil C stocks, these stock changes result from the integration of all management practices undertaken. Change to one management practice often results indirectly in changes to other management practices and the net effect needs to be explicitly understood. This review aims to identify knowledge gaps related to the use and production of supplemental feed, while also providing an overview of the complex nature of soil C change in managed agricultural systems. Additionally, methods for detecting a change in soil C are reviewed including providing the rationale for the use of the net ecosystem carbon balance (NECB) method used in this thesis.

For context, this review begins with an overview of the New Zealand dairy industry (both current and historical) including the role of supplemental feed, which is then followed by the global C cycle. A comparison of flux measurements and direct soil sampling methods is then provided. Finally, the effect of relevant management activities of both grazed grasslands and croplands that influence the results of the studies within this thesis are summarised, including sections covering the current knowledge of supplemental feed use and maize cropping as related to the overarching thesis topic.

2.2 New Zealand dairy industry

In recent years the New Zealand dairy industry has seen considerable expansion and improvement in productivity on both a per cow and per hectare basis. One of the key drivers of improved productivity is the increased use of supplemental feed. As this thesis explores how the use of supplemental feed affects soil C, the following sections firstly describe the expansion of the New Zealand dairy industry and then provide a review of how supplemental feed use has increased.

2.2.1 Changes in the size of the New Zealand dairy industry

In the 20 years between the 1993/94 and 2013/14 milking seasons New Zealand's national dairy herd size increased by 80%, and the land area by 53%, with both stabilising since 2013/14 (DairyNZ, 2019b). This recent increase in herd size has greatly exceeded historical averages, where between 1960 and 1990 the annual increase was about 1.4% per annum (MacLeod and Moller, 2006) compared to 3.1% per annum for the 1993-2013 period. Often intensification is measured in terms of an index of stocking rate (or density) measured as the

number of animals per unit of grassland area. The stocking rate for dairy cattle gradually increased from the 1970s through to 2016 at an average rate of 0.54% per annum, although they may have plateaued during 2013-2019 (MacLeod and Moller, 2006; DairyNZ, 2019b). Higher stocking rates only account for some of the increase in the national herd size with the remainder a result of land-use conversion, primarily from sheep and beef to dairy farming (Journeaux *et al.*, 2017).

While there were increases in both the dairy stocking rate and effective land area, neither can solely account for the recent increases in milk production. Milk production (kg of milk solids (kgMS) produced) increased by 151% between the 1993/94 and 2015/16 (year of peak production) season with an associated per cow productivity increase of 33% (DairyNZ, 2019b). This increase has been in-line with general dairy industry goals of the early 2000s, such as the aim of increasing total productivity by 50% in the 10 years from 2004 to 2014 (Parliamentary Commissioner for the Environment, 2004). Mechanisms for the observed increases in productivity have been primarily through increasing farm inputs, i.e. increased use of fertilisers (both nitrogen and non-nitrogen fertilisers; MacLeod and Moller, 2006) and water (through irrigation), and imported supplemental feed (Foote *et al.*, 2015). Additional genetic gains to the dairy herd, improved farm management and improved pastures, have also contributed to the increased productivity (Bryant and Amer, 2014; DairyNZ Economics Group, 2016).

2.2.2 Supplement feed use

New Zealand's suitable climate with regular rainfall and fertile soils allows for extensive pasture growth, and as such, pasture is the dominant component of the dairy cow diet. However, with the increased intensification and expansion into less climatically suitable locations, supplemental feed has become an increasingly large component of the diet. In 1990/91 pasture was 96% of the average dairy cow diet, while in 2015/16 this was estimated to have reduced to 82% (DairyNZ Economics Group, 2016). The quantity and purpose of supplemental feed use within New Zealand dairy farms vary from an all pasture-based system with no imported feed (i.e. production system 1; DairyNZ, 2019a) through to greater than 50% of feed imported (i.e. production system 5; DairyNZ, 2019a). Supplemental feed (imported and/or on-farm produced) is used to increase milk production by providing feed to lactating cows at times of pasture deficit (feed insurance; Fausett *et al.*, 2015) and/or to support higher stocking rates by feeding year-round, which has generally been shown to increase profitability

(Doole, 2014). The following section separately discusses the use of pasture-based supplemental feeds, and non-pasture (cropped) supplemental feeds, with a summary provided in Table 2.1.

2.2.2.1 **Pasture supplement feeds**

At times of excess pasture growth, often in spring and early summer, the additional pasture is harvested and either dried and stored as hay, or ensiled as pasture silage or balage (ensiled pasture at a higher dry matter content). In New Zealand, pasture-based feed supplements (silage, haylage and hay) are a minor component of the dairy cows diet, which is in contrast to many other countries where such supplements are the basis of the cows' diet (Howse *et al.*, 1996). During the 1990s, pasture silage was the predominant source of supplementary feed on New Zealand dairy farms (MacDonald *et al.*, 2000), with the use of non-pasture supplements limited. In more recent times, despite the increased use of non-pasture supplements, there has also been an increase in the production of pasture silage (Kleinmans, 2016).

Statistics New Zealand (2018b) report 801,882 ha of pasture (including lucerne) harvested for hay, silage and balage for the year to June 2017, of which 48% (388,069 ha) was produced on farms identifying as dairy cattle farming. However, there is a transfer of harvested pasture from non-dairy to dairy farms resulting in difficulty determining the total used by the dairy industry. While the total pasture harvested (measured as dry matter; DM) is more uncertain, assuming a net harvest of 3-4 t DM ha^{-1} (DairyNZ, 2006), an estimated 2.4-3.2 Mt DM in pasture supplements was produced annually. Despite not all of this pasture supplemental feed being consumed by dairy cows, it does provide some context to the comparison with non-pasture supplemental feed (see Section 2.2.2.2). Historical trends are difficult to interpret due to the occasional nature of the surveys (5 yearly), available pasture for harvest due to climatic differences, and previous agricultural surveys amalgamating pasture/lucerne with cereal crops harvested for silage, hay and balage. Nonetheless, in the year to June 2007 about 769,000 ha of pasture was harvested for silage, hay and balage (includes cereal crops; Statistics New Zealand, 2007), ~869,000 ha in the year to June 2012 (Statistics New Zealand, 2012) and ~802,000 in the year to June 2017 (Statistics New Zealand, 2018b) suggesting some consistency in the quantity of pasture supplement produced.

Table 2.1: Summary of key supplemental feeds used in the New Zealand dairy industry. Collated data is from several sources estimated over differing years, but provides an overview of the key supplemental feeds and their relative importance.

Category	Type	Estimated land area	Estimated quantity	Year of estimate	Source
Pasture supplement	Silage	388,069 ha[1]	1,150,000 to 1,550,000 t DM[2]	2016/17	Statistics New Zealand (2018b)
	Haylage				
	Hay				
Non-pasture supplements	Palm kernel expeller (PKE)	n/a[3]	1,643,000 t DM[4]	2019/20	Index Mundi (2020a)
	Maize silage[5]	53,853 ha	1,146,000 t DM	2018/19	AIMI (2019)
	Fodder beet[6]	35,000 ha	698,000 t DM	2014/15	DairyNZ Economics Group (2016)
	Barley[6]	25,680 ha	145,000 t DM	2014/15	
	Kale[6]	73,500 ha	121,000 t DM	2014/15	
	Wheat[6]	7,632 ha	58,000 t DM	2014/15	
	Maize grain[6]	4,950 ha	57,000 t DM	2014/15	
	Swedes[6]	37,200 ha	57,000 t DM	2014/15	
	Cereal whole crop silage[6]	12,000 ha	50,000 t DM	2014/15	
	Turnips[6]	24,000 ha	24,000 t DM	2014/15	

[1] For dairy farms only, therefore ignoring any feed imported from other farm types
[2] Assuming a net harvest of 3 to 4 t DM ha^{-1} (DairyNZ, 2006)
[3] Imported to New Zealand
[4] 1,825 MT imported with an assumed DM content of 90%
[5] Total production, although almost exclusively used in the dairy industry (DairyNZ Economics Group, 2016)
[6] Production for dairy cows only

2.2.2.2 **Non-pasture supplemental feeds**

The most prolific non-pasture feed types are palm kernel expeller (PKE), maize silage and fodder beet (DairyNZ Economics Group, 2016). Importation of PKE to New Zealand has increased from virtually nothing in 1992 (Foote *et al.*, 2015), stabilising at around 1.8 million tonnes between 2018 and 2020 (Index Mundi, 2020a). The 1.825 million tonnes of PKE imported in 2020 equated to 26% of all globally exported PKE (Index Mundi, 2020b). Like PKE, locally grown maize silage (none is imported to New Zealand) has also increased considerably from the early 1990s (4,000 ha sown) through to 2017 where almost 54,000 ha was sown (AIMI, 2019). Along with the increase in the area of maize silage grown, the yield has also increased from ~19 tonnes dry matter per hectare (t DM ha^{-1}) to 22-23 t DM ha^{-1} (DairyNZ Economics Group, 2016). Maize silage growth is also regionally variable with the Waikato region of the North Island accounting for almost 50%, and the North Island accounting for 90% of all maize silage cropping in 2017 (Statistics New Zealand, 2018b). The importance of maize silage in the North Island is mirrored by the importance of fodder beet in the South Island. Fodder beet area sown increased from 50 ha in 1990/91 to 1,000 ha in 2008/09 and 50,000 ha in 2014/15 (DairyNZ Economics Group, 2016). For the 2014/15 season PKE was the largest non-pasture supplemental feed type used accounting for 33% of the total; maize silage accounted for 29%, and fodder beet 16%. No other non-pasture supplemental feeds (such as barley, kale, wheat, etc.) accounted for more than 4% of the total (not reviewed here). Total non-pasture supplemental feed for the 2014/15 season was ~4.5 Mt DM (DairyNZ Economics Group, 2016), a much larger component than pasture supplemental feed (c.f. 2.4-3.2 Mt DM across both dairy and non-dairy sectors for 2017).

2.3 The global carbon cycle and soil C

The global C cycle is based on the interconnection of several pools of C. Lal (2004b) described five principal global C pools (ordered from largest to smallest): oceanic, geologic, soil, atmospheric and biotic. These pools can be further combined (e.g. the soil and biotic combined and described as the terrestrial pool by Janzen (2004)), disaggregated (e.g. Figure 2.1), or often ignored (e.g. the geologic pool) when considering anthropogenic impacts on global C cycling. The exact magnitude of these pools is difficult to quantify accurately, and where estimated, often have large uncertainty; e.g., Friedlingstein *et al.* (2019) estimate the C stock of soils to be between 1500 and 2400 Gt C. Furthermore, the global C cycle is dynamic

and therefore, the size of individual pools of C change through time, whether through natural or anthropogenic processes.

Transfer of C between the differing pools occurs via several mechanisms and over differing timescales. The predominant transfer is between the atmosphere and the other principal pools of C (oceanic, geologic, soil and biotic), but some transfer occurs between the biotic components and soil, and soil and oceanic pools (Lal, 2008b; Galy *et al.*, 2015). Transfers or cycling of C between the atmosphere and the other pools is through the release and/or uptake of carbon dioxide (CO_2) and methane (CH_4) (Janzen, 2004; Saunois *et al.*, 2016). As the focus of this review pertains to soil C, further information on the transfer of C between the oceanic/geologic and atmospheric pools is not presented here but is available in other sources (e.g. Lal, 2008b).

The C cycle of the terrestrial ecosystem begins with the input of C through gross primary productivity (GPP) (Baldocchi *et al.*, 2016). That is, plants take up atmospheric CO_2 through the process of photosynthesis with an estimated 120 Pg C y^{-1} assimilated globally (Lal, 2008b; Baldocchi *et al.*, 2016; Friedlingstein *et al.*, 2019). Approximately half of this photosynthesised CO_2 is released back into the atmosphere through plant respiration (autotrophic respiration), with the remaining half stored in the vegetative tissue (Janzen, 2004). The photosynthesised and stored C (net primary productivity (NPP)) is available for transfer from the biotic C pool to the soil C pool. Heterotrophic respiration, or C lost by organisms in the ecosystem other than plants (Kirschbaum *et al.*, 2001), is typically similar in magnitude to autotrophic respiration (Janzen, 2004; Lal, 2008b). At the global scale, CO_2 uptake (GPP) is generally balanced by respiration (autotrophic and heterotrophic) coupled with losses of plant material through fire (Lal, 2008b; Baldocchi *et al.*, 2016; Friedlingstein *et al.*, 2019). Both photosynthesis and respiration processes are subject to environmental controls including the availability of solar radiation, vapour pressure deficit, temperature and moisture, as well as the physiological activity of the plants (Goulden *et al.*, 1996; Baldocchi *et al.*, 2016). Accordingly, the net CO_2 exchange (the balance of photosynthesis and respiration) at scales smaller than global can be spatially variable and are affected by not only local and/or regional climate but also species composition. Indeed C_4 plant species (including maize) are generally considered to have a photosynthetic rate twice that of C_3 species (Pearcy and Ehleringer, 1984; Kajala *et al.*, 2011), and therefore even under the same environmental conditions, photosynthetic and respiration rates can differ.

While the C cycle is generally a combination of physical and biological processes, human activity plays an increasingly significant role. The most significant anthropogenic interaction with the global C cycle has been the release of C from the geologic pool predominantly through the processes of fossil fuel combustion and cement production (Lal, 2004b), which for the 2009-2018 decade was estimated at 9.5 Pg C y^{-1} (Figure 2.1; Friedlingstein *et al.*, 2019). Of more relevance to this review, land-use change to support human activity has also significantly interfered with the C cycling by modifying natural ecosystems and depleting the soil C pool. Lal (2004b) estimated land-use change through deforestation, biomass burning, conversion of natural to agricultural ecosystems, drainage of wetlands and soil cultivation had resulted in 78±12 Pg C being lost from the soil C pool. Sanderman *et al.* (2018) modelled a global soil C debt due to agriculture of 133 Pg C in the top 2 m of soil. Such losses are still ongoing with Friedlingstein *et al.* (2019) estimating 1.5 Pg C y^{-1} was lost in the 2009-2018 decade through land-use change. Despite these losses of soil C through land-use change and associated agricultural activity, the entire terrestrial land area (i.e. including grasslands, croplands, forests, wetlands, etc.) has historically been a sink of C and was estimated at 3.2 Pg C y^{-1} for the 2009-2018 decade (Figure 2.1; Friedlingstein *et al.*, 2019).

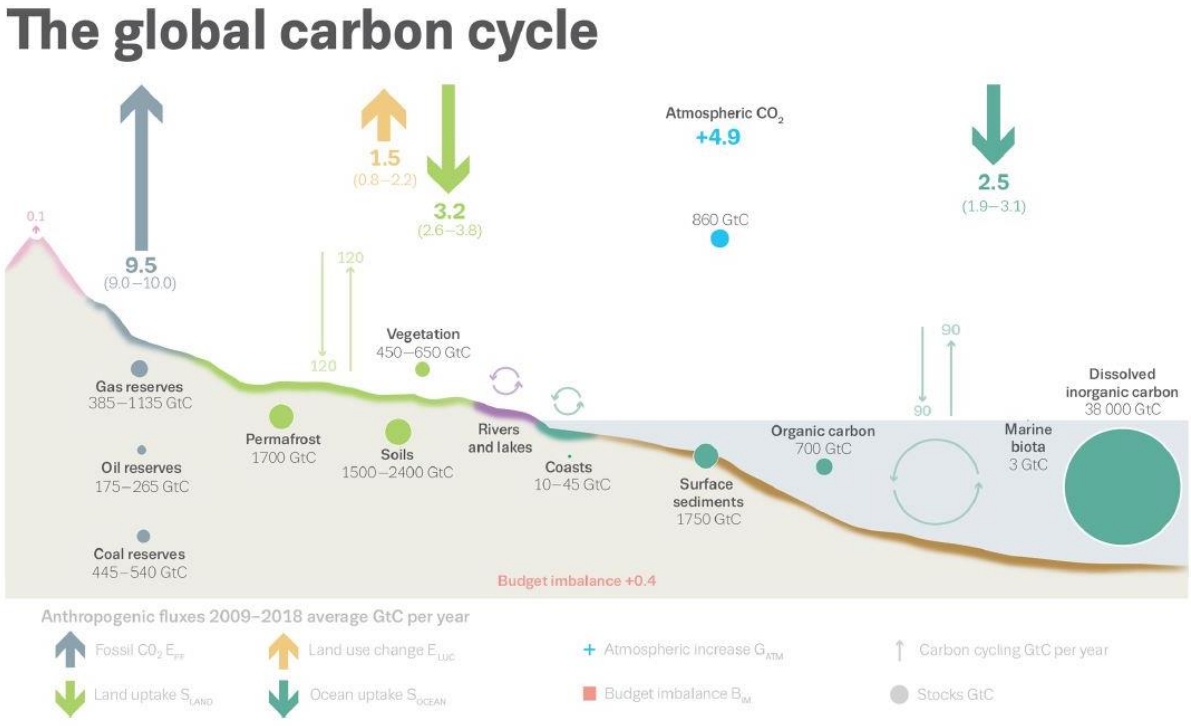

Figure 2.1: The global C cycle averaged for the decade of 2009-2018 indicating the reserves and flows of C (Figure from Friedlingstein et al. (2019)).

Nevertheless, manipulation of the CO_2 exchange processes to allow for increased C stored in vegetation and soil are seen as a possible method to mitigate for increases in atmospheric

CO_2 arising through anthropogenic activity (Janzen, 2004). Options associated with the terrestrial C pool include both management (e.g. reducing emissions) and sequestration of C (Lal, 2008a). Use of the terrestrial ecosystem to mitigate increased atmospheric CO_2 was first proposed more than 40 years ago (e.g. Dyson, 1977), but has gained significantly more relevance in the past couple of decades (e.g. Soussana *et al.*, 2004; Lal, 2008a; Paustian *et al.*, 2019b). This interest resulted in the "4 per 1000 Initiative: Soils for Food Security and Climate" launched at the 21st Conference of Parties (COP21) in 2015 with the aspirational goal of increasing soil C stocks by 0.4% per year (4 per 1000, 2020). While there has been much discussion as to the full plausibility of this initiative (e.g. Poulton *et al.*, 2018; Soussana *et al.*, 2019) the message is clear that there is an opportunity for increased soil C storage to occur and that the debate might be better focused on the total possible storage capacity and management practices needed to realise this enhance storage potential.

Most of the opportunity to sequester atmospheric CO_2 and increase soil C stocks occurs on managed agricultural land (Minasny *et al.*, 2017). Estimates of the sequestration potential of agricultural lands relative to anthropogenic GHG emissions range from 4% through to 15% (e.g. Soussana *et al.*, 2010; Goglio *et al.*, 2015), while Minasny *et al.* (2017) calculate an annual increase of 0.4% within global agricultural soils could offset up to 35% of emissions. These agricultural lands are those that, as previously noted, were primarily responsible for significant losses of soil C through human activity, and thus have considerable potential to recover that lost C. Indeed, soils that have lost the most C, and therefore have low initial stocks, also have the greatest potential (under best management) to have high sequestration rates (Minasny *et al.*, 2017). Consequently, the sequestration potential of soils is influenced by both land use (current and historic) and soil type. For example, long-term croplands are considered to have the potential to sequester much more C than pasture soils (Lal, 2011; McNally *et al.*, 2017).

Anthropogenic activities have modified the global C cycle and the stores in each of the differing C pools, and for example, have reduced the soil C pool by somewhere between 3 and 9% (estimated based on numbers presented in Lal (2004b), Sanderman *et al.* (2018) and Friedlingstein *et al.* (2019)). Considerable research suggests that there are potential for these losses to be recovered through modifying management activities aimed to promote increased uptake of atmospheric CO_2. Significantly, the concept of GHG mitigation also includes reduction of GHG release (Edenhofer *et al.*, 2014), thus includes identifying mechanisms that

minimise losses of soil C (i.e. through the respiratory release of CO_2). Given agricultural soils provide significant potential for GHG mitigation, it is important to develop a thorough understanding of how the management of these lands affect soil C stocks (see Sections 2.5 and 2.6), and therefore can be beneficially manipulated.

2.4 Soil C measurement methods

To evaluate the effect management or indeed any process (e.g. climate change) has on the C cycle requires a method to estimate this change through time. Two common methods for measuring soil C stock change include using direct soil sampling and inferring change from flux measurements (Smith *et al.*, 2020). To detect change through soil sampling requires comparison to a baseline value, often a previous measurement. Traditionally, soil sampling has been the main approach and remains useful for longer time frames and larger scales and is the approach used for inventory purposes (IPCC, 2006, 2019). Inferring change from flux measurements requires an understanding of the type of ecosystem where the measurements are to be made. For example, where flux measurements are made in natural ecosystems, photosynthesis and respiration are responsible for the majority of the flow of C into and out of the terrestrial pool. In the absence of significant leaching and erosion losses of C, quantifying the net CO_2 exchange alone can provide information as to whether there is gain or loss of terrestrial C (Paustian *et al.*, 2019a). However, this may not necessarily correspond to changes in soil C as the change may be in biomass that can contain significant C stocks, especially woody biomass. Agricultural ecosystems differ in that the biomass is generally ephemeral due to annual harvesting and/or grazing (Paustian *et al.*, 2019a), and therefore within these ecosystems, any change in terrestrial ecosystem C is synonymous to the change in soil C. There are advantages and disadvantages to both approaches, and the choice of measurement method depends on the research purpose. In the following section, both approaches are reviewed, with a primary focus on inferred changes through flux measurements as this was the approach used in this thesis.

2.4.1 Soil sampling

In simplest terms, determining the soil C stock via sampling involves taking a soil sample of known volume, and determining the dry mass and C content of the sampled material. However, calculating soil C stocks, and subsequently change in those C stocks is less simple. Methods for determining the C concentration of a soil sample are well established and are

neither technologically challenging nor especially difficult (Paustian *et al.*, 2019a) although can require expensive and operationally intensive equipment. The C content of soils is, however, spatially heterogeneous (e.g. Robertson *et al.*, 1997), and therefore careful experimental design is required (Smith *et al.*, 2020). Consequently, many samples may be required to determine the "average" C stock of an area of interest (Paustian *et al.*, 2019a).

The C content of soils usually has a vertical gradient with a higher concentration in the topsoil decreasing with increased depth and, therefore, the depth of sampling is also an important consideration. For example, IPCC guidelines recommend soil C stocks are computed to a depth of 0.3 m (IPCC, 2006, 2019), while global C stocks are often computed to 1 m (e.g. Janzen, 2004). While much change in soil C stocks occurs in the upper 0.3 m and thus could be accounted for following IPCC guidelines (Paustian *et al.*, 2019a), significant C stocks do exist in the sub-soil below 0.3 m. For example, Soussana and Lemaire (2014) suggest >50% of total soil C stocks could be below 0.3 m depth. Limiting the sampling depth to 0.3 m can be problematic when stock changes are considered where, for example, plough depth is increased in cultivated soils, or deeper rooting plant species are evaluated (Smith *et al.*, 2020). To overcome such limitations, FAO (2019) suggest sampling as deep as possible, while also reporting to 0.3 m for comparability. Indeed, many recent studies also report C stocks deeper than 0.3 m (e.g. Schipper *et al.*, 2014; Mudge *et al.*, 2017).

Comparison of soil C stocks, whether through time or space, can be influenced by differences in sampled soil mass determined through measurement of soil bulk density. Changes in soil bulk density can arise through natural spatial variability, or management differences (e.g. till vs no-till, differing land-uses, etc.), and therefore, sampling to a fixed depth leads to systematic overestimates of C stock in soils with higher bulk densities (Wendt and Hauser, 2013). To correct for this bias, estimation of soil C stocks quantified on equivalent soil mass (ESM) is recommended (Wendt and Hauser, 2013; FAO, 2019; Smith *et al.*, 2020). That is, soil C stocks are calculated on the same mass of soil (representative of the target depth) rather than to the same fixed depth. There are different methods to calculate the C stocks using ESM (Wendt and Hauser, 2013), but the basis is choosing a reference mass appropriate for the target depth, and calculating the C stocks in the same mass of soil from each sample. Consistency of reference mass and that it falls within the sample depth are the important aspects here (Wendt and Hauser, 2013; von Haden *et al.*, 2020).

Quantifying the effect of management on soil C stock using soil sampling methods can be achieved by comparing samples through time following a change in management (e.g. Schipper *et al.*, 2014), using a paired sampling approach (e.g Barnett *et al.*, 2014; Mudge *et al.*, 2017), or using sampling chronosequences by replacing space for time (e.g. Sparling *et al.*, 2014; Carolan and Fornara, 2016). In general, soil sampling to determine C stocks and/or change is challenged by high spatial variability and small changes relative to the C stock (Paustian *et al.*, 2019a). Consequently, a considerable number of samples are required to determine small differences, and if sampling through time, may require several intervening years to detect such changes (Smith, 2004). Furthermore, any changes through time attributed to specific management or treatment need to be compared to an un-altered control rather than just an initial baseline. For example, change in soil C can occur due to drought or other climatic phenomena that are irrespective of the management/treatment (Nayak *et al.*, 2019). Determining the change in soil C stocks through sampling methods involves laborious sample collection and analysis of many samples (Paustian *et al.*, 2019a), but following a consistent sampling protocol forms the basis of much soil C monitoring research (FAO, 2019; Smith *et al.*, 2020) and allows for aggregation across large scale land uses and management.

2.4.2 Flux measurements

An alternative approach to soil sampling involves quantifying all flows (or fluxes) of C into and out of an ecosystem resulting in a C budget (Smith *et al.*, 2020). Chapin *et al.* (2006) proposed the term net ecosystem carbon balance (NECB) to describe the net rate of C accumulation (or loss) in ecosystems, whereby the NECB sums all sinks and sources of C and describes them as a function of time. Similarly, the term net biome productivity (NBP) is also often used, which integrates NECB over time and space (Schulze and Heimann, 1998; Chapin *et al.*, 2006). The C balance approach (whether calculating NECB or NBP) requires the quantification of all flows of C into and out of the ecosystem of interest, with differing flows for differing ecosystems (see below). For the calculated C balance to be synonymous to the change in soil C, the underlying assumption is that C stored as biomass remains constant (Paustian *et al.*, 2019a). Due to this assumption, many studies use the terms C source (net loss of C), C sink (net gain of C) and C neutral (no change) to describe the change in ecosystem C rather than attributing the change entirely to soil C. Additionally, change in C stocks at any depth within the soil

profile is included, therefore removing the depth limitation that can influence reported change determined by soil sampling techniques (Smith *et al.*, 2020)

The key measurement within C balance methodologies is the CO_2 exchange between the atmosphere and ecosystem. This exchange is referred to as either net ecosystem exchange (NEE) or net ecosystem production (NEP), where the two are equivalent in magnitude, but opposite in sign. Positive NECB and NEP are defined by ecologists as an input of C to the ecosystem, while positive NEE is defined by atmospheric scientists as an input of C to the atmosphere (Chapin *et al.*, 2006). Practically, the NECB equation can generically be described as:

$$NECB = -NEE + C_{import} - C_{export} \quad (1)$$

or:

$$NECB = NEP + C_{import} + C_{export} \quad (2)$$

The C_{import} and C_{export} terms vary depending on the ecosystem in question. For example, agricultural systems include lateral flows of product (harvest, milk, meat, etc.) that do not occur in natural ecosystems (Smith *et al.*, 2010; Paustian *et al.*, 2019a). While not an exhaustive list, common import terms used in the NECB equation include synthetic fertilisers, organic amendments (liquid and solid manures, composts, etc.) and supplementary feed. Export terms include gaseous losses (microbially produced methane, volatile organic compounds), groundwater losses of inorganic, dissolved and particulate C, product (milk, meat, harvest), fire, erosion and pollen and seed export by wind (Chapin *et al.*, 2006; Smith *et al.*, 2010). Some of these terms are negligible and are often ignored e.g. leaching losses, erosion, pollen and seed export (Soussana *et al.*, 2007; Paustian *et al.*, 2019a), depending on site location and conditions. For example, because EC measurements are ideally made across flat terrain (Rannik *et al.*, 2012), erosion is often minimal and thus excluded.

To calculate NECB, clearly defined system boundaries need to be defined. That is, the area through which any flows of C are computed and thus defining what is and is not, included within the NECB calculation. Within agricultural ecosystems, the definition of the system boundary is critical, especially when considering material harvested for animal feed (i.e.

supplemental feed) (Smith *et al.*, 2010; Felber *et al.*, 2016a). Smith *et al.* (2010) argue that if crops produced as animal fodder are fed on the farm of production, then the system boundary should be extended to include the livestock and related fluxes. Additionally, respiration of grazing animals themselves can be problematic to NEE and thus NECB calculation (Kirschbaum *et al.*, 2015; Gourlez de la Motte *et al.*, 2019), and therefore determination as to their presence within the system boundary needs to be given consideration (Felber *et al.*, 2016b). In many cases, the system boundary may be simply constrained by the area over which the NEE measurements are representative. Within agricultural ecosystems, this may be <1 to a few hectares (e.g. Ammann *et al.*, 2007; Rutledge *et al.*, 2015), while for natural ecosystems, this may be many hectares (e.g Goodrich *et al.*, 2017; Ney *et al.*, 2019). Regardless, following definition of the system boundary, determination and quantification of appropriate components contributing to the NECB calculation can be made.

Whereas soil sampling methods allow for the calculation of both soil C stocks and if sampled repeatedly through time change in C stock, NECB measurements only allow for calculation of the change in C stocks. However, NECB measurements are advantageous over soil sampling to detect change as they can be made over timescales ranging from a few weeks to annual and longer (Jones *et al.*, 2017; Koncz *et al.*, 2017; Rutledge *et al.*, 2017b). NECB methodology also allows measurement over integrated areas in order of hectare scale and may be able to limit the effect of spatial variability within the area of measurement. Therefore, they are ideal for undertaking treatment or management comparison studies (e.g. Allard *et al.*, 2007; Rutledge *et al.*, 2017a). Undertaking NECB measurements come with limitations including the requirement for expensive EC equipment with substantial specialist knowledge (Eugster and Merbold, 2015), and suitable locations for EC measurements (see below). Furthermore, most studies have minimal or no replication thus limiting interpretation to the study site rather than the ecosystem and/or allowing for generalisation about a treatment or management (Hill *et al.*, 2017).

2.4.2.1 **Measurements of NEE**

Measurements of NEE are predominantly made using either closed chamber or micrometeorological approaches (Riederer *et al.*, 2014). The primary micrometeorological technique used for quantifying NEE is the eddy covariance (EC) technique, which allows for assessment of CO_2 exchange at the ecosystem scale (Baldocchi, 2003). Indeed, ecosystem-scale assessment of NEE makes EC measurements the popular choice for NECB studies,

compared to the small spatial and temporal coverage that chambers quantify (Riederer *et al.*, 2014). Comparison of NEE concurrently measured by chambers and EC have identified differences between the methods attributable to factors including atmospheric conditions (e.g. Riederer *et al.*, 2014) and representativeness (e.g. Reth *et al.*, 2005), and accordingly, Riederer *et al.* (2014) suggest EC be preferable for determination of ecosystem fluxes. Furthermore, Eugster and Merbold (2015) provide a summary table comparing chamber and EC measurements of fluxes, which highlighted larger spatial and higher temporal coverage of flux data obtained by EC – traits that are more beneficial to NECB studies.

2.4.2.1.1 The eddy covariance method

The EC technique measures exchange of heat, mass, and momentum between a flat, horizontally homogenous surface and the overlying atmosphere (Foken *et al.*, 2012). The principle behind the technique is the measurement of the turbulent motions of upward and downward moving air (eddies) and sampling (and subsequent measurement) of the trace gases transported by these motions (Baldocchi, 2003). Fluxes can then be determined by calculating the covariance between fluctuations of the vertical wind speed and the mixing ratio (concentration) of trace gases of interest (Baldocchi, 2008). Measurements are made continuously at rates which are frequent enough to capture the variability due to the atmospheric turbulence (typically >1-10 Hz, but can be faster) (Munger *et al.*, 2012).

Flux measurements are typically made within the surface boundary layer (approximately 20-50 m in the case of unstable stratification, and lower in stable stratification), where fluxes are approximately constant with height (Foken *et al.*, 2012). In this zone, atmospheric turbulence is the dominant transport mechanism making the eddy covariance method suitable for measurements of scalar fluxes. The resultant flux measurements are derived from source areas upwind of the measurement location referred to as the flux "footprint" (Kormann and Meixner, 2001), which is dependent on measurement height, terrain roughness, and boundary layer characteristics (Göckede *et al.*, 2004). The spatial extent over which measurements are made can be manipulated by adjusting instrument height allowing for maximising the coverage of target areas. However, there are several key requirements for an ideal eddy covariance measurement site. Sites need to conform to micrometeorological requirements including adequate fetch for all desired wind directions and atmospheric stabilities (Munger *et al.*, 2012). Additionally, the ideal site includes a topographically flat and even terrain, containing uniform vegetation in the footprint area (Chen *et al.*, 2011; Munger

et al., 2012). In real-world practice, few locations are ideal, and as such, compromises are often required. A general guideline is to have the representative ecosystem contribute >80% to the measured flux (while aiming for >90%) (Munger *et al.*, 2012). Rotationally grazed agricultural systems can be challenging where the flux footprint extends across multiple paddocks, each of which has similar but inherently different management (e.g. timing of grazing) (Kirschbaum *et al.*, 2015) and therefore violates the homogeneity assumption of EC (Rannik *et al.*, 2012). An improved understanding of how to interpret EC measurements obtained from measurement areas extending across rotationally grazed grasslands is required.

2.4.2.1.2 EC flux calculation summary

EC measurements involve the collection of three-dimensional wind data along with trace gas measurements at a high-frequency (typically 10-20 Hz). At a selected time step, often half-hourly, fluxes are processed via a series of calculations, corrections and filtering steps (Burba, 2013). The result is a time series of processed fluxes, containing gaps where data is either missing or of low quality. Filling of these gaps can be achieved by one of several methods including mean diurnal variation, look-up tables, artificial neural networks (ANN), non-linear regressions, or process models (Moffat *et al.*, 2007; Papale, 2012), and is an on-going field of research (although more so for CH_4 and N_2O (Nemitz *et al.*, 2018)). Use of gap-filled or non-gap-filled data is dependent on the purpose. For example, investigation of functional relationships between fluxes and drivers uses non-gap-filled data where more certainty of the measured flux is required. In contrast, aggregation of values to annual sums, such as those required for NECB calculations, must use gap-filled data due to the requirement for values at every time step (Papale, 2012).

2.4.2.2 **Measurement of non-CO_2 fluxes**

Determination of the non-CO_2-C fluxes occurs through measurement, estimation, calculation and a combination of these methods. In many cases, the entire quantity is unable to be measured and requires scaling up from measurements of sub-samples. For example, biomass removed from an entire paddock through harvest is challenging to measure in entirety, and is often randomly sub-sampled and analysed for dry matter and C contents allowing for upscaling to estimate total paddock offtake (e.g. Béziat *et al.*, 2009; Laubach *et al.*, 2019). In grazed systems, estimation of biomass consumed by the animals is even more challenging, and other approaches may be used. Examples include calculations based on per animal intake

(Felber *et al.*, 2016a), and before and after estimates of paddock biomass determined using a calibrated rising plate meter (i.e. measuring herbage height) (Gourlez de la Motte *et al.*, 2016; Hunt *et al.*, 2016). Other components may be estimated using modelling approaches (e.g. DOC leaching; Rutledge *et al.*, 2017a) and book values (e.g. C content of milk (Mudge *et al.*, 2011); grazer CH_4–C emissions (Oates and Jackson, 2014)). Methods for determining the non-CO_2-C fluxes vary from study-to-study depending on both the terms included in the NECB calculation and data availability.

2.4.2.3 **Uncertainty estimates**

NEE uncertainty consists of elements of systematic and random uncertainties (Baldocchi, 2003; Richardson *et al.*, 2012). Richardson *et al.* (2012) summarise the random errors as arising due to (1) the stochastic nature of turbulence and associated sampling errors; (2) errors due to the instrument system; and (3) uncertainty attributable to changes in wind direction and velocity thus affecting the representativeness of the 30-minute measurement. Systematic errors, or bias, arise through (1) unmet assumptions and methodological challenges, instrument calibration and design, and data processing. Generally, the uncertainty associated with NEE is calculated using well established and consistent methods (e.g. Elbers *et al.*, 2011). Uncertainty for differing components are calculated in isolation, and then often propagated by summing in quadrature. For example, Elbers *et al.* (2011) calculated the total uncertainty of NEE as:

$$X_t = \sqrt{x_a^2 + x_b^2 + x_c^2 + x_d^2 + x_e^2} \qquad 3$$

where: x_a was the measurement error and flux uncertainty, x_b was the self-heating correction uncertainty; x_c was u* threshold uncertainty; x_d was the statistical selection uncertainty; x_e was the gap-filling uncertainty. It should be noted that components incorporated in the calculation of NEE uncertainty differ from study-to-study depending on study design and purpose. For example, Gourlez de la Motte *et al.* (2016) include the equivalent of x_a, x_c and x_e from equation 3 (but not x_b or x_d) while also including an uncertainty estimate associated with the choice of cut-off frequency in undertaking spectral corrections of the fluxes. Alternately, as the purpose of Rutledge *et al.* (2017a)'s study was the comparison of nearby treatments, the systematic uncertainties were ignored as they argued each site would have experienced the same systematic uncertainty thus allowing for a decreased detectable difference. Although there are generally standardised methods for calculating the uncertainty

of NEE, there is variability in what is reported between studies (i.e. which components are included) making comparisons challenging. Nonetheless, early on Baldocchi (2003) argued that from nearly ideal sites error bounds are generally less than ±50 g C m^{-2} y^{-1}, a value which is similar to uncertainties reported in many present-day studies (e.g. Barcza *et al.*, 2020; Voglmeier *et al.*, 2020).

The reported uncertainty associated with the non-CO_2 components of NECB, much like the measurements of NEE, is quite variable. Where possible, uncertainty is determined from analysis of sub-samples (e.g. Ammann *et al.*, 2020), otherwise simply estimated (e.g Gourlez de la Motte *et al.*, 2016), while some studies, particularly those reporting many sites and/or years report no uncertainty on an annual basis (e.g. Wang *et al.*, 2015; Jones *et al.*, 2017; Poyda *et al.*, 2019). A more comprehensive approach was undertaken by Rutledge *et al.* (2017a) whereby uncertainty was applied to every measurement involved in the calculations (e.g. for C imported in manure this included uncertainty for mass applied, dry matter and C contents). Uncertainty was then determined using an approach inspired by Bayesian elicitation (O'Hagan *et al.*, 2006) which allowed for a combination of calculated uncertainty and estimated uncertainty based on "expert opinion". Propagation of uncertainty in most studies follows the summing in quadrature approach (i.e. similar to equation 3) (e.g. Mudge *et al.*, 2011; Gourlez de la Motte *et al.*, 2016; Ammann *et al.*, 2020), although Rutledge *et al.* (2017a) calculated 95% probability intervals.

Uncertainties (for both NEE and/or NECB) are reported in a variety of forms including as standard deviations (Schmidt *et al.*, 2012), 95% confidence intervals (Ammann *et al.*, 2020), standard errors (Oates and Jackson, 2014; Eichelmann *et al.*, 2016), 95% probability intervals (Rutledge *et al.*, 2017a), or are un-specified (Hunt *et al.*, 2016). Consequently, it is difficult to interpret NEE/NECB between different studies if uncertainties are included. Use of uncertainties in NEE/NECB studies is more beneficial for determining if a result indicates a C sink, source or neutral, and for comparison of treatments. This summary of uncertainty estimation does, however, identify the lack of standardisation, both in terms of how the uncertainties are calculated and reported, and more so for NECB than NEE.

2.4.3 Summary

Both soil sampling and flux measurements of change in soil C stocks are useful tools for specific investigations. Soil sampling methods are often used to provide average C stocks

and/or change of C stock for land-use (e.g. for inventory purposes (IPCC, 2006, 2019)) through the collection of many samples over a much larger spatial extent. However, such approaches can be very labour intensive, are limited to the depth of sampling (often 0.3 m) and provide minimal temporal information. In contrast, flux measurement based studies can provide considerable temporal detail (although often only for a few years) and are useful for understanding the impacts of a specific site or management mechanism integrated across the area within the defined ecosystem boundary. They are, however, often limited to a single site and lack replication limiting the ability to expand conclusions beyond the ecosystem boundary (Hill *et al.*, 2017). There are also other methods including spectral methods and modelling approaches (Smith *et al.*, 2020), which are not reviewed here. The focus of this thesis is on agricultural management with measurements made using NECB methodology, which was deemed more suitable for studying these specific short-term (<1 to a few years in length) activities. Consequently, the following sections on grassland and cropland management primarily focus on results collected using similar methodology.

2.5 Grazed grasslands

Grazing land area accounts for ~3.4 billion ha globally, storing an estimated 343 Pg C in the top 1 m of soil (Conant *et al.*, 2017). As previously noted, these agricultural lands (i.e. including croplands) are considered to have substantial potential for C sequestration (Minasny *et al.*, 2017; Paustian *et al.*, 2019b), and therefore have received much research focus. However, Conant *et al.* (2017) identified that data on grazing lands had been less widely collected than for forests and cropland, and limited to a subset of regions and management practices. From a C sequestration perspective, grassland research has focused on the effect of management on soil C stocks. Such studies range from broad, covering topics such as the effect of grazing on soil C stocks according to land surface topography, soil type or land-use (e.g. dairy vs dry stock), through to specific management such as stocking rate, fertiliser use and irrigation.

2.5.1 New Zealand grassland soil C stock change

While the effects of specific management practices are discussed below, it is first worthwhile summarising the effects of grazing in a more general sense on soil C stocks, particularly within the New Zealand context where grazed pasture is the predominant land use (Schipper *et al.*, 2017). In recent years several studies have investigated the change in soil C stocks of New Zealand grasslands at decadal scale primarily through repeated samplings (Schipper *et al.*,

2007; Schipper *et al.*, 2010; Parfitt *et al.*, 2014; Schipper *et al.*, 2014). Schipper *et al.* (2017) summarised the findings as changes in soil C stock (to 0.3 m) of flatlands under grazing management being primarily due to soil order rather than grazing type. Allophanic and Gley soil orders were identified as having lost soil C in the past, while the other soil orders sampled (Recent, Brown, Pumice and Pallic) had near-zero change. It should be noted here that the soils of the study site for this thesis primarily consist of Allophanic and Gley soil orders. With limited data, both Schipper *et al.* (2014)and Parfitt *et al.* (2014) suggest that soil C stocks on hill country under pasture increased between samplings, although Schipper *et al.* (2017) note hill country to be highly heterogeneous, and the representativeness of these data sets to be unknown.

Initially, Schipper *et al.* (2010) identified grazing type (dairy vs drystock) to be a predictor of soil C change, but the inclusion of additional samples with an improved representation of soil orders suggested this conclusion might have been at least partly due to unbalanced sampling (Schipper *et al.*, 2014). Using a paired-site sampling strategy, Barnett *et al.* (2014) identified dairy farms to have significantly lower soil C stocks in the A horizon than dry stock farms, although this was not significant for the whole profile (0-0.6 m). Norris (2014) reported similar findings while Houlbrooke *et al.* (2008) found no difference. Schipper *et al.* (2017), however, do caution of the difficulty in assigning differences to grazing type due to the farm systems being management differently (e.g. Barnett *et al.* (2014) identified a ~70% higher stocking rate of the dairy farms than the drystock farms), and therefore differences are a result of all management factors.

2.5.2 Role of grazing animals

Within grazed grasslands, C cycling is intrinsically linked to the grazers themselves. Grazing animals consume biomass C (whether pasture or supplemental feed; C_{Intake}), with most expelled from the animals as either (i) respired CO_2 (C_{Resp}), (ii) released as methane via enteric fermentation (C_{CH4}), (iii) excreted as dung and urine ($C_{Excreta}$), and (iv) exported as milk (C_{Milk}) (Felber *et al.*, 2016a; Gourlez de la Motte *et al.*, 2019). Additionally, a small fraction of C may remain within the animal as its liveweight changes ($C_{Liveweight\ Change}$), which may then be exported from the ecosystem upon slaughter (or death) of the animal (Soussana *et al.*, 2004; Felber *et al.*, 2016a). Each of these terms can be considered as a flux of C, and therefore an animal C budget can be completed, i.e.:

$$C_{Intake} = C_{Resp} + C_{CH_4} + C_{Excreta} + C_{Milk} \pm C_{Liveweight\ Change} \quad 4$$

Of these terms, all except $C_{Excreta}$ result in an export of C from the ecosystem. Consumed C respired as CO_2 and lost as methane are returned to the atmosphere soon after ingestion, and combined can account for more than half of intake C (Rutledge *et al.*, 2017a). Respired CO_2 can be problematic for EC measurements due to the animals themselves being temporary moving sources of CO_2 (Felber *et al.*, 2015; Kirschbaum *et al.*, 2015). Furthermore, the grazing animals may move in and out of the measured EC footprint and therefore violate the assumption of homogeneity required for EC measurements (Rannik *et al.*, 2012). Nonetheless, NECB studies have both included (Allard *et al.*, 2007; Soussana *et al.*, 2007; Rutledge *et al.*, 2015) and excluded (Skinner, 2008; Hunt *et al.*, 2016; Rutledge *et al.*, 2017a) grazers within the defined ecosystem boundary, with recent trends tending to exclude grazers where possible. Exclusion of grazers from the ecosystem boundary treats them as agents of import and export (Hunt *et al.*, 2016) whereby they quickly remove the biomass and export it from the ecosystem boundary, while also importing and depositing excreta. The biomass ingested and thus removed by the grazers includes the proportions allocated to CO_2 respiration, enteric methane release and milk (or product) eliminating the requirement for them to calculated separately, although NEE measured during grazing must be discarded to avoid inclusion of grazer respiration (Hunt *et al.*, 2016; Rutledge *et al.*, 2017a). Felber *et al.* (2016a) reported the C budget of a dairy pasture both including and excluding cows and concluded that the NECB was similar, although noting the large difference in NEE due to the inclusion or exclusion of grazer respired CO_2.

Product exports in the form of either milk or meat are lateral flows of C from the ecosystem. Within dairy farm systems, C in milk exports can account for up to 20% of C intake (Felber *et al.*, 2016a; Hunt *et al.*, 2016). Meat C exports in non-dairy systems meanwhile are generally much smaller than that of milk (Gourlez de la Motte *et al.*, 2016), and is often ignored in NECB calculations (Soussana *et al.*, 2007; Felber *et al.*, 2016a). Estimates of C stored in animal liveweight ranging from 0.136 kg C kg LW^{-1} (Avila, 2006) to 0.165 kg C kg LW^{-1} (Mathot *et al.*, 2012), leading to Felber *et al.* (2016a) estimating C accumulated as liveweight gain by dairy cows to be <2% of milk export (~3 g C m^{-2} y^{-1}) and similar magnitude to findings by Allard *et al.* (2007). Consequently, during short-term grazing events within rotationally grazed systems liveweight gain would be negligible and could be excluded from calculations, particularly for dairy farms where the emphasis is on milk production. However, liveweight change may need

to be included in continuous grazing systems and/or where the main product is meat production.

Remaining C consumed by the grazing animals not otherwise utilised is excreted as dung and urine in the non-digestible fraction. Typically, this is in the range 20-40% of C intake (Allard *et al.*, 2007), and varies according to the quality of the feed. Rutledge *et al.* (2017a) compared their results with five other studies, with the average dung (faeces) deposition of all six studies being 29.5% of C intake. Additional C is excreted in the urine, which Rutledge *et al.* (2017a) report as 14% of dung C. Unlike the gaseous losses (respired CO_2 and methane) and exported product (milk/meat), excreta generally remain within the ecosystem and any decomposition to CO_2 would be captured by flux measurements thus being included in NEE. It is worthwhile noting that grazing animals that move in or out of the ecosystem can transfer dung elsewhere thus modifying this assumption. For example, Mudge *et al.* (2011) calculate the cows spent 1.5 hours per day outside the ecosystem boundary during milking. During this time dung would also have deposited on farm tracks and the milking shed resulting in a net transfer of dung out of the ecosystem boundary, and therefore needs to be accounted for as an export from the ecosystem (Rutledge *et al.*, 2015). Conversely, feed consumed away from the areas of investigation (other paddocks, dedicated feed pad/barn) can represent a transfer of excreta into the ecosystem boundary. To account for such transfers, Felber *et al.* (2016a) assumed excreta was deposited linearly through time and proportioned according to time spent on the paddock. In rotationally grazed systems where grazing durations are short (perhaps as short as a few hours), estimation of excreta deposition is further complicated by the digestive time of the grazing animals. Digestive rates depend on the composition and quality of the feed and can range from a few to almost 50 hours (Moran, 2005). Consequently, in rotational grazing systems with short grazing durations, C grazed within an ecosystem boundary could be entirely exported with the non-digestible fraction excreted elsewhere and deposited excreta consider an import (Rutledge *et al.*, 2017a).

In the context of soil C, grazing animals export the majority of biomass C from the ecosystem whether in gaseous or solid forms, and only that converted to excreta remains. The excreta, however, has the potential to be stabilised as soil C (Section 2.5.3.1) and, therefore, is the main conduit in which animals directly influence soil C stocks (i.e. the conversion of biomass C to excreta C). The quantity of excreta deposited does not necessarily correlate with feed consumed within the ecosystem boundary if additional feed is consumed elsewhere. Finally,

the animal C balance (i.e. equation 4) is a closed-loop that ideally should balance. Depending on the method of calculation, this is not always the case (e.g. Rutledge *et al.*, 2015; Felber *et al.*, 2016a), which may indicate miscalculation of one or components, therefore, affecting interpretation of results.

2.5.3 Management effects

The following sections describe the effect that management practices have on soil C stocks, focussing on those that are frequently used and more relevant to grazed grasslands. Consequently, proposed mechanisms to increase soil C stocks that are also applicable to grazed grasslands (e.g. deep ploughing/flipping, earthworms, dung beetles, etc.) are not covered here. It is also important to note that within grazer based systems the primary purpose of most of these practices is to increase the production of animal feed and thus product (Whitehead *et al.*, 2018).

2.5.3.1 Manure and fertiliser amendments

Manure

Excretion of the non-digestible fraction of consumed animal feed, whether via direct deposition onto the land (i.e. during grazing), or collected and redistributed as manure, provides a source of C available for sequestration in the soil. In a meta-analysis, Maillard and Angers (2014) calculated a global C-retention coefficient of 12% for manure. Given that, approximately one-third of feed intake is excreted (see above), this would suggest that around 4% (⅓ of 12%) of feed C is sequestered in the soil. However, there is considerable variability in the results of individual studies including some that show no gain or even losses (e.g. Angers *et al.*, 2010). Furthermore, sequestration rates of manure depend on the type of animal the manure is derived from, e.g. Maillard and Angers (2014) found cattle manure led to slightly higher sequestration rates than pig and poultry manure. The rate of soil C increases due to manure applications has been positively correlated to the rate of application through both meta-analysis (Maillard and Angers, 2014) and individual studies comparing application rates (Fornara *et al.*, 2020). Moreover, Fornara *et al.* (2020) found a significant increase in soil C stocks only occurred with the application of high rates of cattle manure when comparing low, medium and high rates of cattle and pig manure. Hirata *et al.* (2013) also suggest climate may be important, with larger NECBs following application of manure at sites located in a cool temperate climate compared to one from a warm temperate climate. Finally, Chenu *et al.*

(2019) identified that poor characterisation of manure amendments as a limitation of this field of research, possibly explaining the variability in observed results. Nevertheless, several authors identify the application of manures as one of the main reasons grassland remain C neutral or indeed may be a C sink (Skinner, 2008; Shimizu *et al.*, 2009; Shimizu *et al.*, 2015).

Fertiliser and liming

Application of inorganic (or synthetic) fertilisers is primarily intended to prompt pasture production, and therefore it is often assumed that an increase in aboveground biomass corresponds to increase soil C stocks (Eyles *et al.*, 2015). Indeed, in a synthesis Conant *et al.* (2017) found the application of inorganic fertiliser increased soil C stocks by 0.54 t C ha^{-1} y^{-1} (540 g C m^{-2} y^{-1}). However, Eyles *et al.* (2015) suggest that rather than fertiliser in the general sense, increases in soil C stocks are influenced by nutrient specific fertiliser additions. They summarise that nitrogen (N) fertilisers typically increase soil C stocks, but the application of phosphorus (P) fertilisers do not despite increasing pasture production. Contrastingly, in a comparison of inorganic fertiliser and organic manures, Hirata *et al.* (2013) found the fertiliser only treatments (all applied as N; Shimizu *et al.*, 2013) resulted in a negative NECB (or C source), which contrasted with the manure plots that gained C. However, in the absence of a control plot it is difficult to ascertain if the negative NECB represents a gain or loss of C relative to no application of fertiliser. It does, however, suggest that studies where both manure and inorganic fertiliser are applied together (e.g. Ammann *et al.*, 2009) need to be carefully interpreted before attribution of causes for C gains or losses.

Worth noting is that the application of fertiliser itself can be an import of C into an ecosystem depending on the chemical makeup of the fertiliser (Rutledge *et al.*, 2017a). While the application of N based fertilisers (e.g. urea) may contribute small amounts of C, application carbonate-based lime may introduce significantly more, especially as this is often applied at higher rates. Liming is used to modify soil pH, and thus promote pasture growth, therefore, like the application of N fertiliser may be assumed to increase soil C. Whitehead *et al.* (2018), however, summarised that the impact of liming on soil C stocks is somewhat unclear, while Parfitt *et al.* (2014) found with no clear relationship between soil pH and C stocks in New Zealand soils.

2.5.3.2 **Grazing management**

Grazing vs harvesting

Direct comparisons of the dependence of soil C change on grazing or harvesting of biomass are scarce (Koncz et al., 2017), however, in a review Soussana et al. (2010) identified a higher average C change from grazed systems than harvested and mixed (both harvest and grazing occurring). Two studies which directly compared grazing and harvesting of pasture (Senapati et al., 2014; Koncz et al., 2017) calculated gains in soil C from the grazed systems, but either neutral (Senapati et al., 2014) or losses (Koncz et al., 2017) from the harvested systems. Export of the harvested material was identified as a primary driver in the observed difference, with harvest use efficiency (HUE; the proportion of removed forage to peak biomass) being 40% higher from harvested systems (Koncz et al., 2017). In the grazing systems, a proportion of the consumed biomass is cycled through the grazing animals, and as described in Section 2.5.2, returned via excreta (Senapati et al., 2014) thus explaining the difference in HUE. Oates and Jackson (2014) also compared grazing with harvesting but included two grazing treatments: intensive rotational grazing and extensive continuous grazing with no harvesting for either treatment. The intensive rotational grazing system NECB was greater than the harvest system for both years, however, the extensive continuous grazing system only had a greater NECB for one year, suggesting that generally, grazing was more beneficial than harvesting, but grazing intensity may also play a role (see below). Likewise, Ammann et al. (2007) calculated the NECB for two harvested pastures managed with differing intensities (N application and a cutting frequency) and found the more intensively managed site had a greater NECB. Consequently, both grazing and pasture management play an important role in determining any difference between harvesting and grazing.

Continuous vs rotational grazing

Grazing management can be broadly separated into continuous and rotational grazing systems. In rotational grazing systems, high stocking rates are maintained for short periods followed by a longer duration of biomass recovery with no grazers present (Eyles et al., 2015). Continuous grazing, as the name suggests, involves animals continually grazing on that same land area albeit at a significantly lower stocking rate than rotational grazing. In a review, Eyles et al. (2015) summarised that rotational grazing benefits include stimulation of root turnover, incorporation of surface litter into the soil, and allows time for the plants to recover during

the rest period, with indirect positive effects due to reduced trampling damage. However, there are few direct comparisons of continuous and rotational grazing, and the results of the few studies are somewhat inconclusive. Sanderman et al. (2015) found no difference in C stocks using soil sampling techniques when comparing rotational and continuous grazing. In contrast, using the NECB methodology Oates and Jackson (2014) found a much larger NECB in the rotational grazing treatment. Alemu et al. (2019) undertook a factorial experiment considering both grazing management and species diversity, with soil C stocks greater under rotational grazing plots with lower diversity, and continuous grazing plots with higher diversity, suggesting sward diversity may be important. Additional factors, such as stocking rate (see below), maybe more important than whether the grazing is continuous or rotational, and isolating just the effects of the grazing management is difficult.

Stocking rates

While this section considers the effect of stocking rates, it is firstly worthwhile noting that long-term and/or severe overgrazing is the main mechanism of degradation of soil C stocks in grasslands globally (Dlamini et al., 2016; Whitehead et al., 2018). Stocking rates in grazed grassland systems are typically associated with the intensity of management, and therefore it is difficult to isolate the effect of stocking rate alone. That is, to support higher stocking rates, amendments such as application of N fertiliser are often used, perhaps in conjunction with imported supplemental feed. For example, Allard et al. (2007) calculated greater NECB for a higher (rather than lower) stocking rate system, however, the more intensive management utilised regular inputs of N fertiliser absent from the lower stocking rate (or extensive) system. This finding contradicts the general expectation that higher grazing intensity (i.e. stocking rate) leads to greater soil C loss (McSherry and Ritchie, 2013; Eyles et al., 2015). Eyles et al. (2015) summarise reasons for a greater loss of soil C with higher stocking rates as due to greater removal of aboveground biomass, reduced belowground C inputs through lower root production and higher root litter turnover, and erosion. In a New Zealand study, Barnett et al. (2014) found lower soil C stocks under dairy farms with higher stocking rates when compared with the lower stocking rates of adjacent drystock farms. However, this example again highlights the difficulty in making comparisons, as the farm (dairy vs drystock) and thus animal type (dairy cows vs sheep and beef cattle) and management differ. Finally, McSherry and Ritchie (2013), identified that soil C stocks decreased with increased grazing intensity in C3 plants, but the opposite for pasture dominated by C4 plants, a finding also

supported by Abdalla et al. (2018). Most evidence suggests that higher stocking rates lead to a reduction in soil C stocks, but the effects of associated management will also influence observed changes.

Movement of feed

While the impact of supplemental feed on soil C stocks has not been directly investigated, several authors indirectly note its importance to both grazing management and subsequently C balance calculations. Koncz et al. (2017) report C balances for an entire farm system that included dedicated grazing, mowing and feeding areas. In this system, mowing provided animal feed during winter when grazing was unable to be undertaken and the animals were housed in an open-air corral. The movement of feed from the mowing to the feeding system resulted in a negative NECB in the mowing system, but a positive NECB in the feed system. They also note that the entire farm system (grazing, mowing and feed) was C neutral. Similarly, Oates and Jackson (2014) concluded that imported feed resulted in a rotational grazing system having a favourable C balance, and without that extra input of C attributed to the imported feed the site would likely have been a C source. As summarised by Jones et al. (2017), biomass harvested as animal feed is utilised elsewhere (rather than simply lost) and includes a proportion returned to the grassland as manure and, therefore, influences the C balance at that location.

2.5.3.3 **Pasture management**

Pasture renewal

Pasture renewal (also referred to as renovation, restoration, rejuvenation or reseeding) refers to interventions to a grassland sward to improve its condition, primarily to increase production (Kayser et al., 2018). The general process of pasture renewal involves first killing the existing sward, typically with a herbicide, followed by either cultivation (e.g. ploughing) of the soil and reseeding, or directly drilling seeds into the soil (Rutledge et al., 2017b). Pasture renewal is a common practice within dairy farm systems, and for New Zealand is estimated to be 6-8% of the land area annually (Kerr et al., 2015). The process of pasture renewal has been shown to cause a loss of soil C during the renewal period (Ammann et al., 2013; Rutledge et al., 2014; Rutledge et al., 2017b). Primarily, this loss is attributed to a lack of photosynthetic inputs during the period between killing the existing sward and establishment and the full emergence of the new sward (Willems et al., 2011; Rutledge et al., 2017b; Reinsch et al., 2018).

Enhanced soil respiration during the renewal phase may also be responsible, although this is debated when compared with an undisturbed pasture (Willems et al., 2011; Rutledge et al., 2014; Reinsch et al., 2018). Rutledge et al. (2017b) identified soil moisture content and time the soil remained bare as the key drivers that determined total C losses during pasture renewal, rather than establishment method (e.g. tillage). Few studies examine the longer-term effects on soil C (Ammann et al., 2020), however, two of the three studies doing so showed the year following the pasture renewal to be a C source (Rutledge et al., 2017b; Ammann et al., 2020). Ammann et al. (2020) also reported lower than expected C balances in the two subsequent measuring years, while Rutledge et al. (2017a) suggest one of two sites that had undergone renewal had accounted for the effects of the pasture renewal within three years. Using soil sampling methods, Linsler et al. (2013) reported significant differences in C concentration of topsoil that existed two years after pasture renewal was no longer significant after five years. In contrast to field studies, modelling by Liang et al. (2020) concluded frequent (including annual) pasture renewal may result in gains in soil C. Although substantial evidence indicates the process of pasture renewal results in short-term losses of C, what is less clear is if the lost C is regained and if so, over what period.

Sward type

Choice of sward type may also affect soil C stocks (Eyles et al., 2015). For example, McSherry and Ritchie (2013) report that soil C is not only affected by stocking rate (see above) but also dependent on whether the sward consists of C3 or C4 plants with higher grazing intensity of C4 plants increasing soil C. Increasing sward diversity is also hypothesised to increase soil C stocks (Steinbeiss et al., 2008) but it is unclear whether the causal mechanism is solely due to greater diversity or a greater chance of including species with beneficial and complementary traits. For example, the presence of deeper rooting species in the sward may allow plants access to more water and nutrients than shallow rooting species, thus increasing C inputs and providing a conduit for C to be incorporated deeper in the soil (McNally et al., 2015; Rutledge et al., 2017a). Various studies have tested the impact of increased diversity and inclusion of deeper rooting species (e.g. McNally et al., 2015; Skinner and Dell, 2016; Alemu et al., 2019; Yang et al., 2019), although overall evidence to date has remained inconclusive (Whitehead et al., 2018).

2.5.3.4 **Irrigation**

In regions where rainfall is insufficient to support year-round growth, irrigation is often employed. Whitehead *et al.* (2018) argue that the impact of irrigation can be variable and the results of individual studies can be contradictory. For example in a New Zealand grazing system, Hunt *et al.* (2016) reported that an irrigated grazed pasture had a greater NECB than an unirrigated neighbouring pasture (also grazed), but another study in the same region comparing irrigated and non-irrigated harvested lucerne identified losses from the irrigated site double that of the non-irrigated site (Laubach *et al.*, 2019). Using a paired site soil sampling method, Mudge *et al.* (2017) showed on average irrigated New Zealand soils had lower soil C stocks than adjacent non-irrigated soils, although some individual pairs had greater C in irrigated soils. Whitehead *et al.* (2018) concluded that climate plays an important role, with C stocks in humid climates likely to remain the same or decrease, with increases expected in climates that are more arid where plant productivity is low before irrigation – similar findings to those of croplands (Trost *et al.*, 2013) (see Section 2.6.1).

2.5.3.5 **Summary**

Carbon stocks and their changes in grazed grasslands are a function of the combination of practices employed in the management of the land. Indeed, in many systems modifying one management practice will require a change to another. For example, intensification of management to increase production (be it milk or meat) often involves increasing the stocking rate. However, to provide sufficient animal feed to support a higher stocking rate, changing other processes such as increasing fertiliser use, or importing supplemental feed is likely also required, and the combination of all processes results in soil C stock changes. For this reason, quantifying the effects of individual management practices in isolation is difficult, and a full understanding of the management system (including the role of the grazers themselves) is required.

2.6 Cropping

While the dominant source of feed in New Zealand dairy-based agriculture is through the grazing of pasture, a drive for greater production has resulted in increasing use of supplemental feed grown on-farm or imported from New Zealand farms and overseas. Production of this supplemental feed for use in dairy systems can be via harvesting of pasture, or either grazed or harvested crops. As with the management of grazed grasslands, differing

cropping practices can influence the C stocks of the soils on which they are grown. While the focus of this thesis is on maize as used for supplementary feed, it is worthwhile to firstly provide a more general overview of crop management practices that alter soil C stocks before reviewing the current knowledge on maize.

2.6.1 Crop management effects on soil C

Land-use conversion and cropping duration

Conversion from alternate land uses (e.g. grassland, pasture, forest, etc.) to cropland almost always results in a reduction of soil C stocks (Smith *et al.*, 2016). For example, in a synthesis, Poeplau et al (2011) calculated conversion of grassland to cropland in temperate systems resulted in a rapid reduction of soil C by 36% that stabilised after 17 years to a new lower steady state. While the worlds agroecosystems are all considered to have a depleted soil C pool (Lal, 2011), cropland soils are even more so. Consequently, long-term cropland systems are considered to have considerable potential as a global C sink, through improved management or for larger gains, being converted back to natural vegetation or grasslands (Poeplau and Don, 2015). Indeed, Conant *et al.* (2017) suggest the permanent conversion of cropland to grassland leads to an increase in soil C concentration of 39% (i.e. similar to what Poeplau *et al.* (2011) attributed to being lost when grassland was initially converted to cropland). Moreover, soil C stocks are severely impacted after only a few years of conversion to cropping (e.g. Shepherd *et al.*, 2001). However, Linsler *et al.* (2013) suggest that while a one-off cultivation (or cropping) event of an otherwise pasture site decreases the soil C stocks, these losses may be recovered within a few years after a return to pasture. Indeed, Soussana *et al.* (2010) recommend the introduction of grass leys into cropping rotation as an opportunity to sequester C, with longer leys (more akin to periodic cropping) being more beneficial to C stocks.

Tillage

Tillage is defined as the mechanical manipulation of the soil for crop production (Busari *et al.*, 2015), and can vary from minimally invasive no-till (or zero-till) to full cultivation. While cropland soil C stocks are typically lower than uncultivated grasslands, higher soil C stocks are often found in no-till systems than those utilising increased tillage (Haddaway *et al.*, 2017). Although the effect of decreased tillage intensity may only be small (5%; Bai *et al.*, 2019), a reduction in tillage is seen as beneficial to both the soil and environment, e.g. through a

reduction in erosion (Busari *et al.*, 2015). Indeed, within continuous cropping systems, a reduction in tillage is generally considered to have C sequestration potential (Bernacchi *et al.*, 2005). However, Haddaway *et al.* (2017) note that the majority of studies comparing tillage intensity are limited to the topsoil (limited to 0.3 m or even 0.15 m) where C accumulation is generally observed under reduced tillage intensity, but the effects deeper in the profile are less well known. Depending on the tillage method, C may be shifted deeper in the profile, and shallow sampling may not capture the true change (Olson and Al-Kaisi, 2015). Furthermore, the impact of tillage on soil C can be variable and dependent on other factors (Snyder *et al.*, 2009) including residue management (Blanco-Canqui and Lal, 2009) and initial soil C contents (Haddaway *et al.*, 2017). While no-till systems are generally considered beneficial from a soil C perspective, there are challenges with continuous no-till including poor weed control, stratification of nutrients (including C), risk of compaction, run-off and acidification (Blanco-Canqui and Wortmann, 2020). Consequently, occasional tillage (once every 5-10 years) may be necessary to manage these challenges, which Blanco-Canqui and Wortmann (2020) suggest having a minimal impact of soil C stocks. This conclusion agrees with the summary of Conant *et al.* (2007) that increased frequency of tillage leads to increased soil C losses. In summary, tillage effects on soil C stocks in cropping systems are influenced by intensity and frequency of tillage, although additional factors (i.e. soil, environmental and management) also play a role.

Residue management

Crop residues consist of the plant remnants following harvest (or grazing) and can include components such as the stems, leaves, roots and chaff (Jarecki and Lal, 2003). The quantity of residue remaining depends on the crop purpose. For example, crops for maize silage, or biofuel, likely have little to no remaining residue, while grain only crops may leave significant residue (i.e. the non-grain components, hereon referred to as stover). Additionally, where residues do exist following the harvest of the main crop, they may be used for fodder, fuel or other uses (Lal, 2013). Retention of residue tends to increase soil C (Smith *et al.*, 2008) and is considered an integral component of no-till cropping systems (Lal, 2013). Residues that remain may be either incorporated into the soil through ploughing or left on the surface depending on tillage management (Turmel *et al.*, 2015). Regardless of the method, retention of the residues is considered beneficial for not only soil C stocks but also soil chemical, physical and biological properties related to soil health (Blanco-Canqui and Lal, 2009; Turmel *et al.*,

2015). As residues provide a source of C available to be incorporated into the soil, it is therefore unsurprising that even partial removal of residue can be detrimental from a soil C perspective (Anderson-Teixeira *et al.*, 2009; Blanco-Canqui and Lal, 2009).

Fertilisation

Application of fertilisers, whether inorganic or organic (e.g. manures), are regular amendments applied to croplands to improve soil fertility and prompt growth and thus crop performance. Long-term application of organic manure is deemed beneficial to cropland soil C stocks (Triberti *et al.*, 2008; Chen *et al.*, 2018; Zhang *et al.*, 2018), but practically, regular application of manure may be difficult to achieve through limited availability (Triberti *et al.*, 2008). The impacts of inorganic fertiliser are less clear. Zhang *et al.* (2018) reported increases of soil C relative to a control under fertiliser only treatment (although much less than those including organic manures). In contrast, Blanco-Canqui *et al.* (2014) found fertilisation had little to no effect on soil C, while Congreves *et al.* (2017) showed changes in soil C due to fertiliser additions depended on crop rotation, tillage and measurement depth, thus highlighting the interdependence of various management practices.

Irrigation

Unlike many other management practices, irrigation is essential for crop growth in many regions (Lal, 2004a), and used to improve crop yields in others (Denef *et al.*, 2008). Specific studies comparing irrigated and non-irrigated croplands have found contrasting results. For example, within the same crop field, Gillabel *et al.* (2007) and Denef *et al.* (2008) reported increased soil C under the irrigated area than the non-irrigated area. In contrast, Verma *et al.* (2005) found irrigation led to losses of C compared to a nearby rain-fed only croplands. In a review, Trost *et al.* (2013) concluded that the effect of irrigation is dependent on climate and initial soil C content. Irrigation of croplands leads to increased soil C contents in arid climates with low initial soil C contents, while humid climates with higher initial soil C content show little effect.

Intercropping period

How croplands are managed on an annual basis strongly influence the soil C dynamics of a system. That is, how the land is utilised between the harvest of the main crop in one year and sowing of the main crop in the next year. Options include sowing of a second crop (referred to as double cropping), sowing of a cover crop (or catch crop), or leaving the soil fallow.

Double cropping, such as growing winter-wheat between summer-maize (e.g. Wang *et al.*, 2015), may increase soil C as well as being beneficial for the primary crop yield (Jarecki *et al.*, 2018). Similar to double cropping, sowing of cover crops purposefully covers otherwise fallow land during the intercropping period, but unlike double cropping, the harvest of produced biomass is not the primary motivation. Instead, cover crops are generally incorporated into the soil as green manure before sowing of the next crop and thus none of the sequestered CO_2 is directly exported from the site (Poeplau and Don, 2015). Poeplau and Don (2015) found that cover crop treatments had significantly higher soil C stocks than reference croplands. It should be noted, that cover crops are not necessarily grown for the entire period between cropping, and maybe for just a few weeks of the growing season (e.g. Baker and Griffis, 2005), although this is suggested to have little C sequestration potential (Bavin *et al.*, 2009). A third option is to leave the land bare (or fallow), with this option being least beneficial from a soil C perspective. When fallow, the primary mechanism for soil C change is via soil respiration, which may have a negligible (e.g. Eichelmann *et al.*, 2016) or substantial (e.g. Jans *et al.*, 2010) effect over the intercropping period. While there are several options for the intercropping period, the choice may be restricted by climatic considerations and constrained by the main crop. Regardless, the intercropping period further modifies the soil C stocks of cropland.

2.6.2 Maize cropping

Maize (*Zea Mays* L.; also referred to as corn) is a prominent crop grown worldwide with an estimated production of 875 million tons in 2012 (Ranum *et al.*, 2014). Maize is a C4 plant that is palatable to ruminants and is highly productive in short periods. Uses of maize also include food and industrial products (e.g. starch, sweeteners, oil, beverages, etc.) and ethanol fuel (biofuel) production (Ranum *et al.*, 2014). Consequently, maize has been the subject of many studies including those investigating C dynamics, and in many cases, the focus is on the exchange of CO_2 rather than soil C change (e.g. Wagle *et al.*, 2018; Guo *et al.*, 2019). In this review, the focus is on the impact of maize cropping on ecosystem C stocks (including soil C) rather than just CO_2 exchange, and primarily from results obtained using C balance methods for comparability with this thesis.

The effect of cropland management processes on soil C stock change as outlined in Section 2.6.1 are equally applicable to maize cropping. Consequently, changes in C stocks from maize cropping are the result of the combined effect of management practices including tillage method, the addition of amendments, residue management, etc., and if reported on an

annual basis, the intercropping period (e.g. fallow, cover-crop, or double-crop). Direct comparisons between studies, therefore, becomes difficult, as very few studies are identical. Furthermore, maize cropping often forms one part of longer multi-year crop rotations including soybeans, winter wheat, etc. and spanning two or more years (e.g. Suyker and Verma, 2012; Buysse *et al.*, 2017). Finally, most research has occurred on permanent croplands, and therefore the effect of the maize crop on soil C will be influenced by historical management of the site (Ciais *et al.*, 2011).

Only a limited number of studies have reported C balances of maize cropping systems at paddock (hectare) scale on an annual or crop-season basis. Table 2.2 presents 18 studies where the NECB was determined for cropping systems that included maize, with data covering 29 unique sites, and 49 individual site years (or crop seasons) along with four other studies with a multiple-year average NECB reported. Regardless of management, the majority were a source of C (negative NECB; >80% of site-years), but with a net uptake of CO_2 (positive NEP; 80% of site-years). This summary table provides an overview and generally endorses the conclusions of many studies (see below). However, it also highlights limitations including all studies in Table 2.2 being located in the northern hemisphere, considerable management variation, and although not listed most were from continuous cropland rather than occasional maize production of otherwise grassland.

When compared to other crops, maize production is often identified as the larger source of C. For example, Poyda *et al.* (2019) compared the C balance from 46 site-years of crops in Germany including winter barley, winter wheat, winter rapeseed, grain maize, spring barley and silage maize, and found that silage maize was the largest source of C. From 12 site years, they calculated an average C balance of –428 ± 192 g C m^{-2} y^{-1}, including a largest individual site-year C source of –765 g C m^{-2} y^{-1}. Similarly, Kutsch *et al.* (2010) report a C loss of –417 g C m^{-2} y^{-1} from maize silage site in Italy, which again was a larger source than all other crop types considered. Moreover, individual site-years for the Italian site averaged by Kutsch *et al.* (2010) were presented in Ceschia *et al.* (2010) and included losses of C of up to –645 g C m^{-2} y^{-1}. In both cases, the C balances represent annual totals including the intercropping period where differing winter crops were grown (fennel and ryegrass for the Italian site and a cover crop for the German sites).

Table 2.2: Reported C balances for maize studies measured at the paddock scale using micro-meteorological techniques (eddy covariance, flux-gradient). Intercropping management describes land use between maize crops (if applicable), manure applied indicates whether organic manures were added, and harvest type indicates if only grain were harvested (with stover residues remaining) or all harvestable biomass (i.e. grain + stover) were removed. Net ecosystem production (NEP) and net ecosystem carbon balance (NECB) are in g C m^{-2} with positive values representing a gain by the ecosystem, and negative values a loss.

Location	Year(s)	Reporting period	Intercropping management	Manure applied	Irrigation used	Harvest type	NEP	NECB	Notes	Reference
St Paul, USA	2001-2003	2-year total	Fallow + spring cover crop	No	No	Grain	376	–91	Conventional till	Baker and Griffis (2005)
	2001-2003	2-year total	Fallow + spring cover crop	No	No	Grain	350	–86	Reduced till	
Nebraska, USA	2001-02	Annual	Fallow	No	Yes	Grain	517	7 to 28	Irrigated, continuous maize	Verma *et al.* (2005)
	2002-03	Annual	Fallow	No	Yes	Grain	424	–69 to –50		
	2003-04	Annual	Fallow	No	Yes	Grain	381	–77 to –52		
	2001-02	Annual	Fallow	No	Yes	Grain	529	24 to 42	Irrigated maize-soybean rotation	
	2003-04	Annual	Fallow	No	Yes	Grain	572	45 to 68		
	2001-02	Annual	Fallow	No	No	Grain	510	175	Rainfed maize-soybean rotation	
	2003-04	Annual	Fallow	No	No	Grain	397	100		
Lamasquére, France	2006	Annual	Fallow	No	Yes	All biomass	186	–372		Béziat *et al.* (2009)
North-east Italy	2007	Annual	Fallow	No	Yes	Grain	473	–11		Alberti *et al.* (2010)
	2008	Annual	Fallow	No	Yes	Grain	343	–85		
Cioffi, Italy	2004-05	Annual	Winter crop (ryegrass)	Yes	Yes	Grain	–412	–480		Ceschia *et al.* (2010)
	2005-06	Annual	Winter crop (fennel)	No	Yes	Grain	–274	–519		
	2006-07	Annual	Winter crop (fennel)	No	Yes	Grain	–342	–645		
Klingenberg, Germany	2006-07	Annual	Fallow	Yes	No	Grain	89	–448		

Location	Year(s)	Reporting period	Intercropping management	Manure applied	Irrigation used	Harvest type	NEP	NECB	Notes	Reference
Langerak, Netherlands	2005-06	Annual	Fallow	Yes	No	Grain	–271	–496		
Winnipeg, Canada	2006-07	Annual	Fallow	No	No	Grain	72	–51		Glenn *et al.* (2010)
	2007-07	Annual	Fallow	No	No	Grain	–7	–314		
	2008-07	Annual	Fallow	No	No	Grain	240	–48		
Dijkgraaf, Netherlands	2007	Crop season	n/a	Yes	No	All biomass	597	–102		Jans *et al.* (2010)
Weishan, China	2008	Crop season	n/a	No	Yes	Grain	244	–124		Lei and Yang (2010)
Dresden, Germany	2007	Annual	Fallow	No	No	n.s.	–136	–199		Prescher *et al.* (2010)
Tongyu, China	2003	Annual	Fallow	No	Yes	All biomass	44	–74	Irrigation only during sowing	Qun and Huizhi (2013)
	2004	Annual	Fallow	No	Yes	All biomass	3	–69		
	2005	Annual	Fallow	No	Yes	All biomass	62	–54		
	2006	Annual	Fallow	No	Yes	All biomass	6	–83		
	2007	Annual	Fallow	No	Yes	All biomass	–55	–101		
	2008	Annual	Fallow	No	Yes	All biomass	33	–65		
	2009	Annual	Fallow	No	Yes	All biomass	–7	–93		
Yucheng, China	2003-2012	10-year average	Winter wheat	No	Yes	Grain	476	–76		Bao *et al.* (2014)
Luancheng, China	2007-2012	5-year average	Winter wheat	No	Yes	Grain	13	–564		
Hebei, China	2008	Crop season	n/a	No	Yes	Grain	143	–167		Wang *et al.* (2015)
Ontario, Canada	2014	Annual	Fallow	Yes	No	All biomass	–64	–699		Eichelmann *et al.* (2016)
Lonzée, Belgium	2012	Crop season	n/a	No	No	All biomass	600	–160		Buysse *et al.* (2017)
Shouyang, China	2012	Annual	Fallow	No	No	Grain	491	25		Gao *et al.* (2017)
	2013	Annual	Fallow	No	No	Grain	553	61		

Location	Year(s)	Reporting period	Intercropping management	Manure applied	Irrigation used	Harvest type	NEP	NECB	Notes	Reference
	2014	Annual	Fallow	No	No	Grain	484	24		
Ontario, Canada	2012	Annual	Fallow	Yes	No	Grain	400	93		Sulaiman *et al.* (2017)
	2013	Annual	Fallow	Yes	No	Grain	64	–380		
	2014	Annual	Fallow	Yes	No	Grain	100	–173		
Piotou-Charentes, France	2011	Crop season	n/a	No	No	All biomass	29	–368		Senapati *et al.* (2018)
Baden–Württemberg, Germany	2010	Crop season	n/a	Yes	No	All biomass	284	–212	Site EC1	Poyda *et al.* (2019)
	2013-14	Annual	Cover crop	Yes	No	All biomass	394	–332	Site EC1	
	2013-14	Annual	Cover crop	No	No	All biomass	436	–174	Site EC2	
	2010-11	Annual	Cover crop	No	No	All biomass	–33	–765	Site EC3	
	2014-15	Annual	Cover crop	No	No	All biomass	205	–448	Site EC3	
	2016-17	Annual	Cover crop	Yes	No	All biomass	294	–295	Site EC4	
	2010-11	Annual	Cover crop	Yes	No	All biomass	119	–732	Site EC5	
	2014-15	Annual	Cover crop	Yes	No	All biomass	398	–302	Site EC5	
	2009-10	Annual	Cover crop	Yes	No	All biomass	272	–250	Site EC6	
	2012-13	Annual	Cover crop	Yes	No	All biomass	167	–561	Site EC6	
	2015-16	Annual	Cover crop	Yes	No	All biomass	106	–611	Site EC6	

The impact of maize harvest type, i.e. grain only vs. full harvest for silage/biofuel, was highlighted by Eichelmann *et al.* (2016). They presented C balances for a Canadian biofuel maize crop separating grain only and full harvest (i.e. grain + stover). The stover component accounted for almost half the crop biomass (~300 g C m^{-2}), and removal of this component substantial increased the C source of the site from –393 g C m^{-2} y^{-1} to –699 g C m^{-2} y^{-1}. While not a direct comparison, Poyda *et al.* (2019) also report one site-year of maize grain with a much lower calculated C loss (–122 g C m^{-2} y^{-1}) than for maize silage (average –428 g C m^{-2} y^{-1}) again suggesting stover removal to increase the C source strength by ~300 g C m^{-2}. The two comparisons highlight the importance of stover removal in calculating the C balance from maize cropping. Furthermore, following a change from grain only to full harvest (grain + stover) removal, using soil sampling rather than NECB, Anderson-Teixeira *et al.* (2009) identified decreased soil C stocks throughout the profile (although proportionally greater in the topsoil) further supporting and emphasising the role of stover removal/retention has on soil C in maize systems. While stover removal in maize cropping systems depends on the purpose (i.e. grain vs silage/biofuel) rather than choice, retaining the stover as residues lead to increased soil C stocks relative to removal, and beneficially has also been shown to increase crop productivity (He *et al.*, 2019; Wang *et al.*, 2019; Liu *et al.*, 2020). Finally, analysis of the data presented in Table 2.2 found a significant difference ($p < 0.05$) between annual NECBs from maize cropping sites with full biomass harvest (average NECB of –365 g C m^{-2} y^{-1} from 10 sites) and grain only (average NECB of –174 g C m^{-2} y^{-1} from 15 sites). However, there was no difference in NEP further emphasising the importance of the additional stover removal to the C balance for maize cropping systems (Figure 2.2).

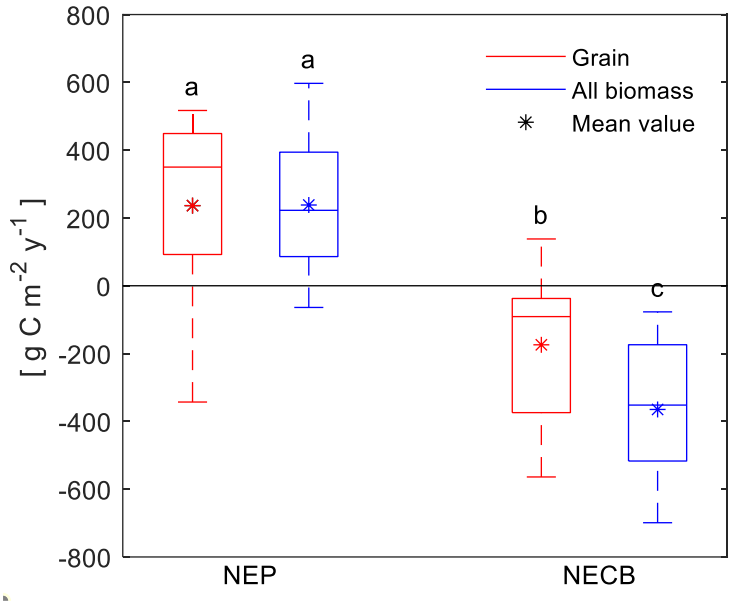

Figure 2.2: Comparison of NEP and NECB for grain only and all biomass harvest type. Data is taken from Table 2.2 with site-averages calculated where multiple years were reported (i.e. one data point per site). Only studies where annual totals (including the intercropping period) were reported were used (25 total study sites).

Few studies report the C balance of just the maize cropping period, but rather include the intercropping period (see examples in Section 2.6.1). This is important because the management of the intercropping period can significantly influence the annual C balance. For example, Wang *et al.* (2015) separated the C balance for the two crops of a winter wheat-summer maize rotation and reported the maize crop (grain only harvest) to be a C source of –167 g C m^{-2} while finding the winter wheat crop to be a C sink of 90 g C m^{-2}. Consequently, for that site, the winter-wheat crop mitigated some of the losses from the maize, and reporting solely the annual total (–77 g C m^{-2} y^{-1}) would have minimised the effects of the maize crop. Similarly, Jans *et al.* (2010) calculated a C balance of –102 g C m^{-2} during the maize cropping phase (maize + stover removal), but including the fallow winter period, this increased to –365 g C m^{-2} y^{-1}. These examples highlight the challenge of determining the effect of solely the maize crop on when data is presented annually.

As noted earlier, most studies where the effect of maize cropping on soil C is investigated are undertaken on long-term continuous croplands and, therefore, the reported findings may not be representative of changes found in the first few years of maize cropping of recently previous grasslands or forest. This is especially important because soil C is likely vulnerable to a rapid loss in the first few years following conversion to cropping (Poeplau *et al.*, 2011). While most studies report losses of ecosystem or soil C from maize cropping (e.g. >80% of site years

in Table 2.2), Verma *et al.* (2005) and Gao *et al.* (2017) identified gains (ranging from 7 to 175 g C m^{-2} y^{-1}) albeit from sites which had been cropped for many years and only removed grain. In contrast, following the conversion of permanent grassland to maize cropping, Struck *et al.* (2020) reported much larger losses of 13 t C ha^{-1} (1,300 g C m^{-2}) for the first two years utilising conventional tillage practices. Although not C stocks, Sparling *et al.* (1992) found organic C % decreased with time under continuous maize cropping across several New Zealand soils following conversion from pasture. Shepherd *et al.* (2001) expanded on the work of Sparling *et al.* (1992) and showed that while total C losses generally increased, the rate of soil C loss slowed as time under maize cropping increased. These studies indicate the length of time under cropping following a conversion may be important for interpreting soil C change under maize cropping, and substantial losses may occur in the first few years.

Maize cropping, and indeed all cropping, results in lower soil C stocks. Factors influencing the soil C stocks are a combination of those that influence CO_2 exchange of the crop (e.g. crop type, soil type and climate) coupled with management (Poyda *et al.*, 2019). For maize systems, keeping stover on-site can, in some cases, halve observed losses, and the addition of organic manure can partially offset such losses. Additionally, irrigation can further modify maize crop C dynamics (e.g. Verma *et al.*, 2005), while C balances reported annually include the effect of the intercropping period with this management being variable from site-to-site. In summary, maize cropping tends to lead to losses in soil C, which can be substantial, although modification due to a wide variety of management makes comparisons between sites very difficult.

2.7 Summary and identification of research gaps

Use of supplementary feed within New Zealand dairy farming systems is increasing (DairyNZ Economics Group, 2016), particularly through the use of imported feed. This imported feed represents a large quantity of introduced C to a farm system, which modelling predicts will lead to an increase in soil C (Kirschbaum *et al.*, 2017). However, prior to this thesis, there was little experimental information tying imported supplemental feed (and embodied C) to increased soil C. Cycling of C (as feed) within grazing animals results in a proportion of that C available for sequestration following excretion (Felber *et al.*, 2016a), which scales with feed intake. Therefore, an increase in imported supplemental feed leads to an increase in excreta, and thus C available for incorporation into the soil. Sequestration coefficients for manure (excreta) of around 12% (Maillard and Angers, 2014) suggest that eventual inclusion of the

imported supplemental feed C into the soil may be low (~4%), but positive, and therefore gains in soil C would be expected following the use of imported supplemented feed.

While gains at the location of use could be expected, consideration needs to be given to the effect of the production of supplemental feed on soil C at the location of production. One of the common forms of supplemental feed in New Zealand is maize silage (DairyNZ Economics Group, 2016), which internationally is the crop most detrimental to soil C stocks (Poyda *et al.*, 2019). However, due to historical land management coupled with present-day crop management the impact of maize silage production within New Zealand dairy farm feed production systems may not be comparable with international research. Indeed, most studies originate from the northern hemisphere, and there is scant information on soil C change from periodic maize silage cropping – a prominent practice in New Zealand (Booker, 2009). An evaluation is required of the system-wide impact of supplemental feed (whether imported or made on-farm) in New Zealand dairy farming systems on soil C stocks.

Estimation of short-term (e.g. annual) changes in soil C stocks is best achieved via the NECB method (Chapin *et al.*, 2006) rather than repeated soil sampling, that generally, cannot accurately measure these relatively small changes in short periods. Use of the NECB method, particularly within grazed ecosystems, is still challenging and further refinement of the methodology is required. For example, improved understanding of the interpretation of fluxes and C balances obtained across multiple paddocks in rotationally grazed systems would be beneficial (Kirschbaum *et al.*, 2015). Additionally, the definition of the system boundary within a farming system is rarely considered and is especially important where the movement of feed and/or use of dedicated feed systems are involved (Smith *et al.*, 2010; Felber *et al.*, 2016a; Koncz *et al.*, 2017). Therefore, while a useful method, improved understanding of the limitations and interpretation of results obtained utilising the NECB approach is required.

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Chapter 3:

Carbon budget of an intensively grazed temperate grassland with large quantities of imported supplemental feed

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Please refer to Appendix D for the contribution of each of the authors

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Carbon budget of an intensively grazed temperate grassland with large quantities of imported supplemental feed

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ABSTRACT

Management of agricultural grasslands can alter soil carbon (C) stocks, and quantifying the effects of different management regimes is important to inform global greenhouse gas (GHG) mitigation strategies. Here, we report the net ecosystem carbon balance (NECB) over three years for an intensively managed rotationally grazed New Zealand dairy farm importing 11.6 t DM $ha^{-1} y^{-1}$ (526 g C $m^{-2} y^{-1}$) as supplemental feed. We investigated whether importing large quantities of supplemental feed to increase milk production led to a gain in soil C. Eddy covariance measurements were used to quantify net ecosystem productivity (NEP), with all other imports and exports of C calculated from farm records and measurements. Additionally, we compared the NECB calculated with a system boundary equivalent to the farm boundary ($NECB_{Farm}$), to that calculated with the system boundary around the paddocks in which the EC footprint extended ($NECB_{Footprint}$).

Both the three-year average $NECB_{Farm}$ and $NECB_{Footprint}$ were similar, but could not be distinguished from zero (71 ± 77 g C $m^{-2} y^{-1}$ and 56 ± 77 g C $m^{-2} y^{-1}$, respectively; uncertainties are standard deviations). The farm boundary approach made better use of available C import and export data (excluding NEP) data, however, error propagation resulted in almost identical uncertainties of the C balance for both methods. Supplementary feed was the major C input of $NECB_{Farm}$ but was only a minor component of $NECB_{Footprint}$ due to the use of a dedicated feed pad outside of the $NECB_{Footprint}$ boundary. In contrast, transfer of C from excreta (including that resulting from the supplementary feed consumed on the feed pad) was the major C input of $NECB_{Footprint}$.

Consumption of the supplementary feed by grazing animals resulted in most of the imported C exiting the system boundary via pathways that included the export of product (milk), respiration of CO_2 and methane production. Only the imported supplementary feed C excreted by the grazing animals along with the C in any feed that was wasted was available for sequestration. When coupled with manure C retention coefficients only a small gain in soil C could be expected. Our result demonstrated that importation of a large quantity of supplemental feed C did not lead to a large gain in soil C. Production of supplemental feed off site likely contributed to considerable GHG production and full life-cycle assessment is needed to determine the overall consequence of increasing milk production with supplemental feed.

1. Introduction

Globally, the agriculture sector is estimated to contribute 10–12% of all greenhouse gas (GHG) emissions (Smith et al., 2014). While agricultural activities emit nitrous oxide (N_2O) and methane (CH_4), they may be either a source or sink for carbon dioxide (CO_2) (Tubiello et al., 2015) which can be represented by changes to the soil carbon (C) stocks. The C sink potential of soils has prompted much interest in identifying mechanisms that lead to increases in soil C as a mitigation strategy for global GHG emissions (Minasny et al., 2017). Most of the potential to increase soil C is on managed agricultural lands (Conant et al., 2017; Minasny et al., 2017), including managed pastures. Managed or grazed pastures cover approximately 26% of the ice-free terrestrial surface (Steinfeld et al., 2006), and are suggested to have soil C sequestration potential that may attain 4% of global GHG emissions (Soussana et al., 2010). Accordingly, quantifying the management effects of these grazed pastures on changes in soil C stocks, and identifying those which provide mitigation potential is of increased importance.

Measurement of changes in soil C stocks through time can be

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difficult, especially when the time scale is short (e.g. annually). The net ecosystem carbon balance (NECB) methodology provides an indirect measurement of the changes in soil C stocks by quantifying all C fluxes at the boundary of the ecosystem under investigation (Chapin et al., 2006). Utilising NECB methodology allows assessment of inferred changes in soil C at timescales ranging from a few weeks to annual and longer (e.g. Jones et al., 2017; Koncz et al., 2017; Rutledge et al., 2017b) and allows determination of key components that contribute to the full carbon balance. Unlike natural (unmanaged) ecosystems, where the C balance is dominated by the exchange of CO_2 between the atmosphere and ecosystem, managed pastoral systems have several additional flows of C into and out of the ecosystem. Flows of non-CO_2-C include both imports (e.g. fertiliser, manure, supplemental feed), and exports (e.g. pasture and supplemental feed removal by animals, and mechanical harvesting), with the location of the (eco)system boundary determining the components included in the C balance (Felber et al., 2016a; Gourlez de la Motte et al., 2016). Central to the calculation of the C balance in grazed systems is the influence of the C cycle of the grazing animals.

The NECB methodology allows for quantifying the effects of pastoral management activities, often on an annual basis, and at hectare and larger spatial scales. Studies have investigated management activities including grazing intensity (Allard et al., 2007; Klumpp et al., 2011), rotational and continuous grazing (Mudge et al., 2011; Oates and Jackson, 2014; Rutledge et al., 2014), fertilisation and manure application (Ammann et al., 2007, 2009; Shimizu et al., 2009; Limin et al., 2015), cut and carry via mechanical harvesting (Zeeman et al., 2010; Koncz et al., 2017), and increasing plant species diversity (Rutledge et al., 2017a), amongst others. At annual timescales, gains, losses and neutral C balances have been reported. The few paired site NECB studies showed a general trend of more intensive management (e.g. rotational grazing and higher stocking densities) resulting in larger gains (or smaller losses) of C than less intense management (e.g. cut and carry, and lower stocking density) (Allard et al., 2007; Klumpp et al., 2011; Oates and Jackson, 2014). Contrastingly, other non-NECB studies suggest otherwise. Higher stocking densities may only result in soil C gains under certain pasture species (McSherry and Ritchie, 2013), and have the potential to lead to soil C losses as a result of increased erosion (Eyles et al., 2015). Additionally, increased use of irrigation to promote pasture growth, and thus grazing intensity, can lead to soil C losses (Mudge et al., 2017).

In New Zealand, the dairy industry has undergone considerable expansion in the past quarter-century, with development into regions not previously associated with dairying, and increased productivity of existing dairy systems. Mechanisms for the observed expansion have been primarily attributed to increasing farm inputs, i.e. increased use of fertilisers (both nitrogen and non-nitrogen fertilisers; MacLeod and Moller, 2006), water (through irrigation), and imported supplemental feed (Foote et al., 2015). While the effects of irrigation and fertiliser on soil C stocks have been studied (Shimizu et al., 2009; Limin et al., 2015; Mudge et al., 2017), measurements of the effect of increased supplemental feed on soil C stocks are scarce.

Kirschbaum et al. (2017) modelled a scenario exploring increased use of imported supplemental feed on milk production and soil C stocks. In their simulation, about 4% of additional C was sequestered in soil C over a 50-year period, although this rate was higher (> 7%) in the first few years following the initial increase in supplemental feed usage. The C within the supplemental feed consumed by the grazing animals is either lost as respired CO_2, emitted as methane, exported in product (meat/milk), or excreted in urine and dung (Gourlez de la Motte et al., 2016). As such, the only C from this process that has the immediate potential to be sequestered by the soil is that in the excreta portion. Additionally, a component of the supplemental feed that is wasted during the storage, supply and consumption process is also available for potential sequestration (typically between < 5% if fed on a dedicated feed pad, and > 20% if fed on the pasture; DairyNZ, 2017). Maillard and Angers (2014) reported a soil C retention coefficient of 12% for farmyard manure (i.e. excreta), and assuming approximately one third of all animal C intake is excreted (Rutledge et al., 2017a), this would also suggest approximately 4% of supplemental feed C could be sequestered as soil C.

The aim of this study was to determine the C balance (NECB) for a New Zealand dairy farm where imported supplemental feed was a large component of the grazing animals' diet, and thus a large net import of carbon to the system. To our knowledge, this is the first study to put constraints on the carbon addition via imported supplemental feed using NECB methodology. The measurement farm imported supplemental feed accounting for > 40% of the grazers diet during each of the three observed years (1 June 2013 to 31 May 2016). Eddy covariance measurements of CO_2 flux were coupled with measurements and literature values for non$-CO_2$ components of the NECB to estimate soil C change. To make the best use of available data, we also evaluated the choice of two ecosystem boundaries on the NECB calculation. The NECB was calculated for both an area constrained by the footprint of the eddy covariance measurements, as well as the area constrained by the farm boundary.

2. Methods

2.1. Site description

This research was undertaken on a commercial dairy farm (Ryan Farm) in the Waikato region of New Zealand (37° 45' 56" S, 175° 46' 00" E, 53 m a.s.l). The dairy farm was rotationally grazed with a sward of mainly perennial ryegrass (*Lolium perenne*) and has been under this vegetation and management for more than 100 years. The predominant soil of the farm was the Waihou silt loam, classified as a Typic Orthic Allophanic Soil (Hewitt, 1998). The Waihou silt loam is a well-drained soil formed from rhyolitic ash on rhyolitic alluvium parent material and is found on flat to undulating parts of the landscape (McLeod, 1992). The carbon content of the soil Ap horizon was 8.3% (McLeod, 1992), bulk density was 780 kg m^{-3} (LandcareResearch, 2017) and the permanent wilting point (PWP) was 0.24 $m^3 m^{-3}$ (McLeod et al., 2016). Mean annual temperature was 13.3 °C, and mean annual precipitation was 1250 mm based on 30-year (1981–2010) averages from a nearby climate station (NIWA, 2018). All precipitation was from rainfall with occasional light frosts occurring during the winter months.

Table 1
Supplemental feed imports to the farm, pasture growth, total herd size and milk produced for each year. Supplemental feed imports and pasture growth are in t DM y^{-1}, total herd size in number of cows, and milk produced in million litres of milk. Year 1 was from 1 June 2013–31 May 2014; Year 2 was from 1 June 2014–31 May 2015; and Year 3 was from 1 June 2015–31 May 2016.

Year	PKE (t DM y^{-1})	Maize Silage (t DM y^{-1})	Maize Husk (t DM y^{-1})	Grass Silage (t DM y^{-1})	Total Imports (t DM y^{-1})	Pasture Growth (t DM y^{-1})	Total Herd Size (cows)	Milk Produced (10^6 litres)
1	700	600	n/a	n/a	1300	2001	510	2.42
2	700	500	39	n/a	1239	1781	525	2.67
3	1000	600	n/a	6	1606	1611	535	2.77

2.2. Farm management

Ryan Farm was rotationally grazed year-round by Friesian-Jersey crossbred dairy cows (500 kg liveweight). Herd size increased in each of the three years from 510 cows in Year 1 to 535 cows in Year 3 (Table 1). The farm was 119 ha with an average stocking rate of 4.40 cows ha^{-1}, equivalent to 3.7 LU ha^{-1} (assuming 1 LU (livestock unit) of 600 kg). The regional average stocking rate was 2.95 cows ha^{-1} (DairyNZ, 2016). Most of the paddocks were nominally 3 ha in size (2.9–3.1 ha), with the remainder (contributing < 5% of the effective area) consisting of smaller sized paddocks that were infrequently grazed by the cows. Three separate herds grazed the farm, with each herd containing an equal number of cows and receiving the same grazing area each day. Grazing rotation length (i.e. the time between grazing events) varied throughout the year from approximately 14 days in the spring-summer period to > 60 days in winter. The length of time the herds remained on each paddock varied from 1 day in the spring-summer period to several days during winter, at which time daily stocking rates were up to 325 cows ha^{-1}. Each day the cows received a prescribed amount of dry matter consisting of pasture and supplemental feed. At times of the year where less pasture was available, the cows received an increased quantity of supplemental feed. The grazable area was also adjusted to ensure that supplemental feed was received by the cows each day, even in times with sufficient pasture available. When excess pasture was available, this was harvested and ensiled for use as supplemental feed in times of lower pasture growth. Supplement feed consisted of predominantly palm kernel expeller (PKE) and maize silage, with smaller quantities of grass silage and maize husk (Year 2 only). All PKE, maize silage, and maize husk were imported to the farm (Table 1), while most grass silage was made on the farm (i.e. not imported). With the exception of the PKE, which was imported from South East Asia, all other imported supplemental feeds were sourced locally (and in the case of maize silage within a few kilometres). All PKE and maize silage were fed on a dedicated feed pad immediately prior to milking rather than on the paddocks. Because of the imported supplemental feed, milk production (Table 1) was less affected by climatic variability and responded to increases in herd size and total feed availability.

Nitrogen (in the form of urea) was regularly applied (several times each year) resulting in total N inputs of about 200 kg N ha^{-1} y^{-1} from fertiliser. No other fertilisers were applied during the measurement period. Collected effluent was applied to the entire farm annually, spread across three different applications (each covering approximately one-third of the farm).

2.3. Instrumentation

2.3.1. Eddy covariance instrumentation

The net CO_2 exchange (reported here as net ecosystem production (NEP); of the same magnitude, but opposite in sign to net ecosystem exchange (NEE)) measurements were made using the eddy covariance (EC) technique (Baldocchi, 2003). The EC system consisted of a 3-D sonic anemometer (CSAT3; Campbell Scientific Inc., Logan, UT, USA), and a LI-7500 A open path infrared gas analyser (LI-COR Inc., Lincoln, NE, USA). Measurements were made at 20 Hz, using a CR3000 datalogger (Campbell Scientific Inc.) and stored on a compact flash card. The instruments were mounted at 2.72 m height and located on the boundary between two paddocks near the centre of the farm (Fig. 1). Footprint analysis (Kormann and Meixner, 2001) revealed that 87% of the measured flux originated within the farm boundary, with 14 paddocks (approximately one-third of the farm) contributing > 0.5% to the final flux.

2.3.2. Meteorological and soil instrumentation

Ancillary meteorological measurements included air temperature and relative humidity measured at 2.65 m (HMP155; Vaisala, Helsinki, Finland), incoming shortwave solar radiation and net radiation at 1.05 m (NR01; Hukseflux Thermal Sensors, Delft, Netherlands), and rainfall at 0.45 m (TB3; Hydrological Services). Volumetric soil moisture content (CS616; Campbell Scientific Inc.) and soil temperature (107 type probes; Campbell Scientific Inc.) were measured at 0.05 m, 0.10 m and 0.20 m below the surface. Additionally, soil temperature was measured with a four-junction averaging thermocouple (TCAV; Campbell Scientific Inc.) where two probes were located at each of 0.02 m and 0.06 m depth below the surface (referred to as 0.04 m soil temperature henceforth). Soil heat flux (HFP01; Hukseflux) was measured at 0.08 m depth. All soil and meteorological measurements were made at 1 Hz and averaged or totalled every 30 min. Data were recorded on either a CR3000 or CR1000 datalogger (Campbell Scientific Inc.).

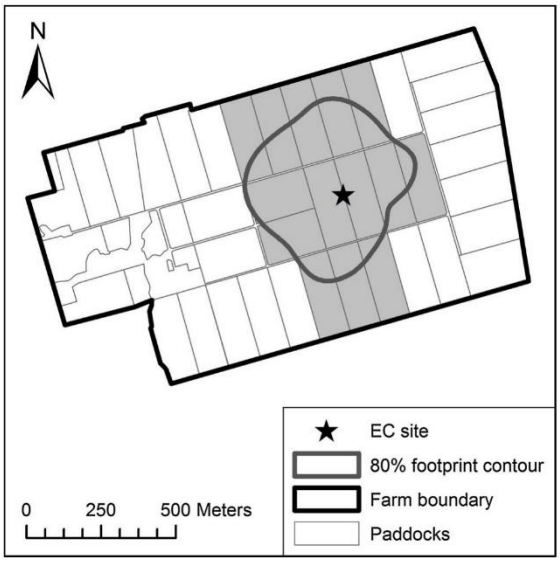

Fig. 1. Map of Ryan Farm. All paddocks within the farm contributed to $NECB_{Farm}$, while only those shaded in light grey contributed to $NECB_{Footprint}$. The location of the EC site is identified by the black star, and the 80% cumulative footprint contribution the thick grey line (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article).

2.4. Flux computation and data processing

Fluxes were computed from the 20 Hz raw data at half hourly intervals using EddyPro (EddyPro ®, 2016). Raw data were screened for spikes, amplitude resolution, drop outs, values above or below absolute limits, and skewness and kurtosis (Vickers and Mahrt, 1997). Double-axis coordinate rotation was used for tilt correction. Data were detrended using block averaging, covariance maximisation was used for timelag determination, and concentrations converted to mixing ratios following Burba et al. (2012). Fluxes were corrected for high-pass (Moncrieff et al., 2004), and low pass filtering effects (Moncrieff et al., 1997) using a fully analytical approach (confirmed as being suitable by examination of sensible heat cospectrum). The software also performed a quality check using the 0-1-2 system developed by Mauder and Foken (2004). Final CO_2 fluxes were calculated as the sum of the turbulent flux and the storage term (Foken et al., 2012).

Following calculation, the half-hourly CO_2 flux data were then subjected to a series of filtering criteria whereby data were rejected when: (i) either the sonic anemometer or LI-7500 A reported warnings equating to greater than 0.5% of the half hour; (ii) calculated fluxes were unrealistically high (NEP > 36 μmol m^{-2} s^{-1}) or low

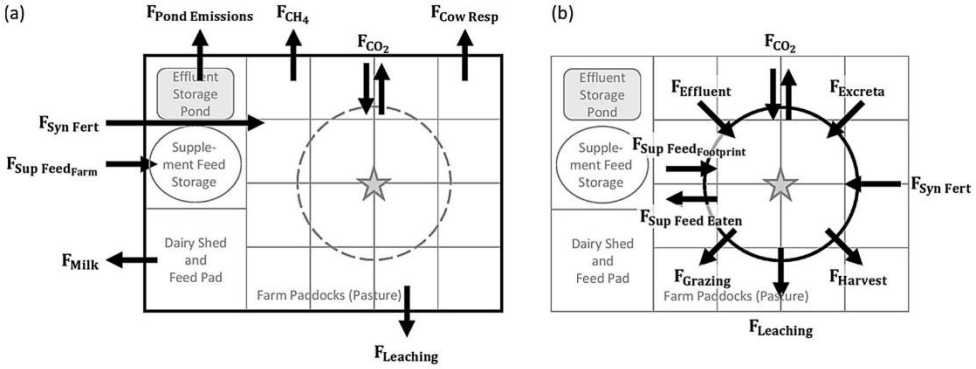

Fig. 2. Schematic illustrating the different system boundaries used to calculate $NECB_{Farm}$ (a) and $NECB_{Footprint}$ (b). The system boundary (thick black line) in (a) borders the farm boundary, while in (b) is constrained by the EC footprint (also shown as the dashed grey line in (a)). NECB component fluxes and their direction are shown with the black arrows. The grey star indicates the EC site (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article).

(NEP < –18 μmol m^{-2} s^{-1}); (iii) the quality check flag had a value of 2; (iv) fluxes calculated under conditions of low turbulence as determined by low values of the standard deviation of the vertical wind speed (σ_w; Acevedo et al., 2009), where the σ_w threshold of 0.11 m s^{-1} was determined by adapting the methodology of Reichstein et al. (2005); (v) the standard deviation of the wind direction was either too large (σ_{dir} > 40) or too small (σ_{dir} < 1) (Hunt et al., 2016); (vi) the source area for the flux came from behind the supporting tower; and (vii) < 80% of the day-time, and < 70% of the night-time flux footprint originated from within the farm boundary. We also deliberately discarded half hours where the flux footprint originated from paddocks occupied by cattle (Hunt et al., 2016; Rutledge et al., 2017a). Known animal location and timing of movement (from grazing records) was combined with the proportion of the flux originating from each paddock on the farm (using flux footprint modelling; Kormann and Meixner, 2001) for each half hour. Any half-hour where > 5% of the flux originated from a paddock that contained grazing cattle was rejected. As a result, an additional 8–10% of the data was rejected annually. Finally, fluxes were also rejected if the modelled flux value calculated by the gap-filling method deviated strongly from the measured flux (Rutledge et al., 2017a). Following filtering 31.8%, 36.1% and 32.7% of the data remained for each of the three measurement years respectively. The energy balance ratio of the three years was 0.84, similar to a nearby grassland study (Pronger et al., 2016).

Gaps in the NEP flux time series were filled using an artificial neural network (ANN; Papale and Valentini, 2003). Day- and night-time data were filled separately. Input variables for the night-time ANN included air temperature, 0.04 m and 0.10 m soil temperature, 0.05 m and 0.10 m volumetric soil moisture content, ΔNEE, and horizontal wind speed. The day-time ANN used incoming shortwave solar radiation, vapour pressure deficit, air temperature, 0.04 m soil temperature, 0.10 m volumetric moisture content, and ΔNEE as input variables. ΔNEE is the difference between daily averaged night-time NEE and daily averaged NEE during day-time non-light-limiting conditions. Non-light-limiting conditions were defined as PPFD > 700 μmol m^{-2} s^{-1} except for the months of May to August (winter) where PPFD > 500 μmol m^{-2} s^{-1} was used. ΔNEE is the step prior to calculation of the phytomass index (PI) first used by Aurela et al. (2001) and Lohila et al. (2004) (where PI is equal to ΔNEE divided by the maximum ΔNEE of a given period). ΔNEE was used as a proxy for the quantity of biomass (photosynthetic material) in the measurement area and has been found to be particularly useful for gap-filling purposes in rotationally grazed systems where rapid biomass change can occur (Campbell et al., 2015).

To partition NEE into gross primary production (GPP) and ecosystem respiration (ER) we first expanded the night-time ANN to run on day-time input variables. The resultant modelled day-time CO_2 flux was assumed to be equivalent to ER for the daytime. GPP was then calculated as the difference between NEE and the modelled daytime ER. GPP was assumed to be zero during the night, and therefore the measured and gap-filled night-time NEE was concluded to represent night-time ER. We used the sign convention where both ER and GPP are positive values. Note that both NEP and ER exclude the respiration of grazed cattle.

2.5. Carbon balance measurements

The net ecosystem carbon balance (NECB) calculation followed the principles of Chapin et al. (2006) in accounting for all sinks and sources of C within an area defined by a system boundary. Generally, the system boundary (where EC is used to measure the CO_2 flux component of the NECB) is related to the area over which the EC measurements are made (e.g. Rutledge et al., 2015; Hunt et al., 2016; Gourlez de la Motte et al., 2018). Here, we refer to this as $NECB_{Footprint}$. The system boundary for $NECB_{Footprint}$ was defined as the paddocks overlapping the 80% cumulative footprint contribution (as calculated from the Kormann-Meixner footprint model (Kormann and Meixner, 2001); Fig. 1). In this study, we also used an expanded system boundary to include the entire farm (referred to as $NECB_{Farm}$), with the single point EC measurement of the CO_2 flux assumed to be representative of the farm. While the calculation of $NECB_{Footprint}$ required all flows of C into and out of each paddock to be known, expansion of the system boundary to incorporate the entire farm only required measurement of the C imported and exported from the farm (Fig. 2). However, the $NECB_{Farm}$ approach also required the following assumptions: (1) grazing management was the same across the farm; and (2) the soils were uniform throughout the farm. The first assumption was valid for Ryan Farm because all paddocks were the same size, the total milking herd was separated into three herds of the same size, and all herds received the same amount of daily feed (pasture and supplement). The soils were mostly similar for the entire farm, with only small pockets of different soils in some locations. It should also be noted that we did not include fossil fuel emissions associated with farm vehicles or operation of the dairy shed in this study.

$NECB_{Farm}$ (Fig. 2a) was calculated as:

$$NECB_{Farm} = F_{CO_2} + F_{Syn\ Fert} + F_{Sup\ Feed_{Farm}} - F_{Cow\ Resp} - F_{CH_4} - F_{Milk} - F_{Leaching} - F_{Pond\ Emissions} \quad (1)$$

Where: F_{CO_2} is the CO_2 flux as NEP (excluding grazer respiration); $F_{Syn\ Fert}$ the C imported in synthetic fertilisers; $F_{Sup\ Feed_{Farm}}$ the C imported

in supplement feed to the farm; $F_{Cow\ Resp}$ the C respired as CO_2 from the cows (excluded from the F_{CO_2} term); F_{CH_4} the C emitted as methane from the cows; F_{Milk} the C exported as milk; $F_{Leaching}$ the C leached below 60 cm soil depth; $F_{Pond\ Emissions}$ the C lost as CO_2 or CH_4 from the anaerobic effluent storage ponds.

$NECB_{Footprint}$ (Fig. 2b) was calculated as:

$$NECB_{Footprint} = F_{CO_2} + F_{Syn\ Fert} + F_{Sup\ Feed_{Footprint}} + F_{Excreta} + F_{Effluent} - F_{Grazing} - F_{Sup\ Feed\ Eaten} - F_{Harvest} - F_{Leaching} \quad (2)$$

Where: F_{CO_2}, $F_{Syn\ Fert}$ and $F_{Leaching}$ are the same fluxes as those used in calculation of $NECB_{Farm}$; $F_{Sup\ Feed_{Footprint}}$ is the C imported in supplement feed into the paddocks within EC footprint; $F_{Excreta}$ the C in excreta returned to the paddocks; $F_{Effluent}$ the C reapplied to the paddocks as effluent from the effluent storage ponds; $F_{Grazing}$ the C in the pasture grazed by the dairy cows; $F_{Sup\ Feed\ Eaten}$ the C in supplement feed consumed by the grazing cattle within the EC footprint paddocks; and $F_{Harvest}$ the C in the pasture mechanically harvested from the paddocks within the EC footprint.

2.5.1. Animal C cycle

We determined how much C was ingested by the grazing cattle and then how this was partitioned in order to enable calculation of a number of key C fluxes. Following Felber et al. (2016a) a C budget considering the losses, gains and storage components for the grazing dairy cows can be constructed as:

$$C_{Intake} = C_{Cow\ Resp} + C_{CH_4} + C_{Milk} + C_{Excreta} \pm C_{Meat} \quad (3)$$

C ingested as feed (C_{Intake}) can be lost as respired CO_2 ($C_{Cow\ Resp}$), emitted as methane from enteric fermentation (C_{CH_4}), converted to milk (C_{Milk}), and excreted in dung and urine ($C_{Excreta}$). The final term (C_{Meat}) is equivalent to a storage term whereby C may be stored or lost depending on the weight change of the animal. Felber et al. (2016a) determined C_{Meat} to be < 2% of the milk C yield and considered the term negligible. We considered C_{Meat} to be zero on an annual basis.

To aid in conceptual understanding, all feed C received by the cows (pasture, grass silage, maize silage, maize husk, and PKE) was partitioned into the components in Eq. (3) for each type of the feed. C_{Intake} was calculated for each feed type as:

$$C_{Intake} = DM \times \%C \times utilisation \quad (4)$$

where: DM was the total dry matter available; %C was the C content of the feed type; utilisation was the proportion consumed by the animals (i.e. to account for pasture/supplemental feed trampled and wasted). The C contents of the supplemental feeds were measured, while the pasture C content was estimated to be 45% based on measurements made on a nearby farm (Rutledge et al., 2017a). We used utilisation values of 0.85 for pasture (Macdonald et al., 2008), 0.8 for supplemental feed consumed on the paddock, and 0.95 for supplemental feed consumed on the dedicated feed pad (DairyNZ, 2017).

C_{Milk} was calculated as the total milk volume exported (as reported by the milk processing company) multiplied by milk density (1.035 kg L^{-1} at 4 °C) and C%. Milk C% was estimated by an equation developed from 12 months of measurements on a nearby farm grazing similar dairy cows, and was:

$$Milk\ C\% = 0.6909 \times MS\% + 1.66 \quad (5)$$

where: MS% is the milk solids percentage reported by the milk processing company. Proportioning C_{Milk} to each feed type was based on the contribution that feed type made to the total feed mass.

$C_{Excreta}$ is the sum of the C in the excreted dung (C_{Dung}) and urine (C_{Urine}). The dung component of the ingested C was calculated as:

$$C_{Dung} = C_{Intake} \times (1 - digestibility) \quad (6)$$

where: digestibility is the percentage of the organic matter able to be assimilated by the animal. Digestibility was calculated from measurements (for supplementary feed) or estimates (for pasture; from: DairyNZ, 2017) of metabolisable energy (ME) by rearranging the equation ME = 0.16 * digestibility (Hill Laboratories, 2019) to: digestibility = ME / 0.16. See supplementary material Table S1 for digestibility values used. C_{Urine} was assumed to be 14% of C_{Dung} (Rutledge et al., 2017a).

Assuming C_{Meat} to be zero, the remainder of the C ingested (i.e. $C_{Intake} - C_{Milk} - C_{Excreta}$) was assumed to be released as respiration ($C_{Cow\ Resp}$) or methane (C_{CH_4}). A survey of several studies where all components from Eq. (3) were reported (Crush et al., 1992; Pinares-Patino et al., 2009; Rutledge et al., 2015; Felber et al., 2016a; Hunt et al., 2016; Rutledge et al., 2017a), provided average contributions (of ingested C) of 52.9% for respiration and 3.9% for methane. We used this ratio to partition respiration from methane.

2.5.2. Components common to $NECB_{Farm}$ and $NECB_{Footprint}$

Eqss (1) and (2) used identical values for the F_{CO_2}, $F_{Syn\ Fert}$ and $F_{Leaching}$ terms. F_{CO_2} is NEP, as described in Section 2.4, and where a positive value represents a gain of C by the ecosystem, again noting that grazer respiration is excluded. $F_{Syn\ Fert}$ is the C imported in synthetic fertilisers applied to the farm. The only fertiliser applied during the study period was urea, with an assumed C content of 20% based on its chemical formula. The imported C was calculated as the mass of applied fertiliser multiplied by 0.2.

$F_{Leaching}$ is the C lost as dissolved organic carbon below 0.6 m depth. Drainage was estimated from the Woodward model (Woodward et al., 2001), using inputs of evaporation (as measured by the EC system) and rainfall. A DOC concentration of 3.7 µg C mL^{-1} was used following measurements made on a nearby farm with soils of the same complex (Sparling et al., 2016).

2.5.3. Components only relevant to $NECB_{Farm}$

$F_{Sup\ Feed_{Farm}}$ is the C imported to the farm from supplement feed. The mass of each type of supplement feed imported to the farm was recorded by the farmer as a total mass of dry matter. Samples of each supplement feed were analysed for carbon content, and the total C imported in supplement feed was calculated by multiplying the total mass (dry weight) by the fractional C content.

F_{Milk}, $F_{Cow\ Resp}$ and F_{CH_4} were calculated following the procedure outlined in Section 2.5.1 for all feed that the grazing cattle received (pasture and supplement feed).

$F_{Pond\ Emissions}$ is the C lost (as CO_2 and CH_4) from the anaerobic effluent storage ponds. Excreta deposited on the feed pad and in the milking shed, along with any wasted supplemental feed from the feed pad, was washed as farm dairy effluent (FDE) into a storage pond. The FDE was stored until being spread back onto the pasture. During the storage period, anaerobic decomposition of the organic matter produces biogas containing both CH_4 and CO_2. $F_{Pond\ Emissions}$ was calculated as:

$$F_{Pond\ Emissions} = PE_{CH_4} + PE_{CO_2} \quad (7)$$

Where PE_{CH_4} is the C from the CH_4 biogas component and PE_{CO_2} is the C from the CO_2 component. PE_{CH_4} was calculated as:

$$PE_{CH_4} = Wastage_{CH_4} + Dung_{CH_4} + Urine_{CH_4} \quad (8)$$

Both $Wastage_{CH_4}$ and $Dung_{CH_4}$ were calculated using the generic equation:

$$X_{CH_4} = VS_X \times EF_X \quad (9)$$

where: X is either wastage or dung; VS_X is the volatile solids of the component; EF_X is the CH_4 emission factor of the component. Emission factors of 0.22 m^3 CH_4 kg^{-1} VS for dung (Heubeck et al., 2014) and 0.31 m^3 CH_4 kg^{-1} VS for feed waste (Chung et al., 2013) were used. The volatile solids were calculated as the total solids multiplied by 0.79 for dung and 0.958 for feed waste (Chung et al., 2013). Total solids for dung was calculated by multiplying the total dung produced by the percentage of time spent on the feed pad and milking shed. The total

dung produced was calculated as outlined in Section 2.5.1, and the time spent on the feed pad and in the milking shed was calculated to be 11% annually. Total solids for supplement feed wastage was calculated as the total supplement feed available multiplied by a wastage proportion (equal to 1 – utilisation). We assumed 5% wastage (W. Ryan pers. comm. 2016). $Urine_{CH_4}$ was estimated as 14% of $Dung_{CH_4}$ as for the determination of C_{Urine} in Section 2.5.1.

Finally, PE_{CO_2} was calculated by multiplying PE_{CH_4} by a typical CO_2:CH_4 ratio of FDE biogas. A biogas composition of 70% CH_4 and 30% CO_2 was used (McGrath and Mason, 2004).

2.5.4. Components only relevant to $NECB_{Footprint}$

$F_{Grazing}$ is the pasture C removed by grazing cattle. Pasture in the paddock at the time of grazing was determined by estimating pasture growth rates based on regular measurements of pasture height using a rising plate meter. A standard calibration of pasture height (in cm) * 140 + 500 (DairyNZ, 2008) was used to estimate the biomass in the paddocks. From repeated measurements, an average daily growth rate for all paddocks on the farm was calculated. On occasions where measurements were not made, growth rates were estimated by using growth rate data from a nearby farm with continuous pasture growth measurements (Rutledge et al., 2017a). All daily growth rate data were then summed between grazing dates to give the standing biomass in the paddock prior to grazing. $F_{Grazing}$ was then estimated as the standing biomass > 4 cm (grazing height) multiplied by a pasture utilisation value of 0.85 (Macdonald et al., 2008), and the C content of the pasture (45%).

$F_{Harvest}$ is the pasture harvested and ensiled. The mass of pasture harvested was estimated by summing up the daily growth rates from the period between the previous grazing and date of harvest for each paddock considered part of the EC footprint (Fig. 1). The mass of pasture was then multiplied by the carbon content of the pasture (45%).

$F_{Sup\ Feed_{Footprint}}$ is the supplement feed imported to the paddocks within the EC footprint. While most supplement feed (all PKE and maize silage) was fed on the dedicated feed pad, all grass silage and maize husk were fed on the grazed paddocks, and thus entered the $NECB_{Footprint}$ system boundary. Total grass silage C included that harvested on the farm and imported from external sources. Given the assumption that the entire farm was managed uniformly, we assume the paddocks in the EC footprint were representative of the entire farm, and thus the $F_{Sup\ Feed_{Footprint}}$ flux was calculated as the total C import (grass silage and maize husk) to the paddocks divided by the farm area.

$F_{Sup\ Feed\ Eaten}$ is the supplement feed imported to the EC footprint paddocks which was consumed by the grazing animals. $F_{Sup\ Feed\ Eaten}$ was calculated as $F_{Sup\ Feed_{Footprint}}$ multiplied by the proportion ingested by the grazing cows (0.8; DairyNZ, 2017).

$F_{Excreta}$ is the C in dung and urine deposited on the paddocks of the EC footprint. Total excreta C from all feed sources was calculated (as described in Section 2.5.1), then proportioned to the time spent on the paddocks (determined to be 85%). All paddocks on the farm were assumed to have the same usage allowing the flux to be calculated as the total excreta divided by the effective area of the farm.

$F_{Effluent}$ is the C in effluent reapplied to the paddocks following emptying of the storage ponds. The total C applied to the paddocks was calculated as the total C entering the effluent storage ponds minus that lost as emissions of CH_4 and CO_2 (i.e. as calculated for $F_{Pond\ Emissions}$). Similar to $F_{Excreta}$, the total C entering the effluent storage ponds was calculated as the total excreta multiplied by the percentage of time the cows spent on the feed pad and in the milking shed. The effluent ponds were emptied two or three times per year, with the entire farm receiving effluent once per year. We, therefore, assumed the total C applied to the paddocks was applied uniformly to all paddocks, and thus dividing by the effective area of the farm gave $F_{Effluent}$.

2.6. Uncertainties

2.6.1. CO_2 flux uncertainties

Uncertainties associated with the eddy covariance measurements include both random and systematic uncertainties (Richardson et al., 2012). The random uncertainty (σ_r) associated with the measurement and gap-filling process was determined using the method outlined by Dragoni et al. (2007). Additionally, the systematic uncertainty related to the choice of σ_w threshold (σ_{turb}) for identifying and rejecting half hourly fluxes where turbulence was poorly developed was determined. Using a method similar to that used by Elbers et al. (2011) for the uncertainty associated with the choice of u^* threshold, we calculated the annual NEP at two additional σ_w thresholds 0.05 m s^{-1} above and below our chosen threshold of 0.11 m s^{-1} (i.e. σ_w = 0.06 and 0.16 m s^{-1}). Uncertainty associated with the choice of σ_w threshold was calculated as the standard deviation of the three NEP totals. The total NEP uncertainty (σ_{NEP}) was calculated following random error propagation rules as:

$$\sigma_{NEP} = \sqrt{\sigma_r^2 + \sigma_{turb}^2} \quad (10)$$

2.6.2. Non-CO_2 flux uncertainties

Direct measures of the uncertainty associated with the non-CO_2 flux components were scarce as measurements were often singular (e.g. digestibility measurements), reported by the farmer (e.g. total supplement feed imported), or required the use of information derived from other sources (e.g. utilisation). We applied an estimated uncertainty to all terms of the equation(s) used to derive each NECB component. The uncertainty of the NECB component was then calculated using random error propagation rules. For example, the import of supplemental feed was calculated as the total dry matter imported multiplied by the carbon content, and thus uncertainties were applied individually to the dry matter import and carbon content terms and then propagated. The uncertainties used were equivalent to an estimate of the standard deviation of each term, and thus equitable to the uncertainty calculated for NEP. To assign uncertainties we initially determined the plausible range for each term. Assuming a normal distribution, and that the plausible range covered 95% of the values, the uncertainty was calculated as one quarter of the plausible range, i.e. equivalent to one standard deviation. Where measurements were available, the plausible range was small (e.g. ± 5% for measured C content), while the range was much larger for reported values (e.g. ± 50% for all emission factors used in the estimation of the pond emissions flux uncertainty). Uncertainty (as a percentage) for each term are reported in Table S2. Finally, NECB uncertainty (σ_{NECB}) was calculated by summing the square of each error term (including σ_{NEP}) and then taking the square root of the sum (Gourlez de la Motte et al., 2016).

3. Results

3.1. Weather

Annual rainfall for the three years averaged 1073 mm (Table 2) and was below the 30-year normal of 1250 mm calculated from a nearby weather station (NIWA, 2018). Year 1 had the lowest rainfall total of 987 mm (Fig. 3a), with well below normal rainfall in the January-March period (29% of normal rainfall in this period). Years 2 and 3 had very similar annual rainfall (1127 and 1104 mm, respectively), although with different monthly rainfall patterns. For July to March of Year 2, monthly rainfall was always below the monthly normal, and well below in the January and February months. Conversely, in Year 3 all months except October, December and April were close to, or above, average rainfall. Dry summers in Years 1 and 2 resulted in periods of each year where the volumetric moisture content (VMC) of the soil at 10 cm depth fell below the permanent wilting point (PWP; 0.24

Table 2
Annual and 3-year averages for the three measurement years from Ryan Farm, and 30-year normal rainfall and air temperature from a nearby weather station for the period 1981–2010. Each year each spanned from 1 June to 31 May of the following year. Table 2a: Environmental variables: annual rainfall (PPT), annual air temperature (T_A), average air temperature for the January to March period ($T_{A\ Jan\text{-}Mar}$), and the number of days the volumetric moisture content at 10 cm was below permanent wilting point (PWP) (VMC < PWP). Table 2b: CO_2 flux components: ecosystem respiration, excluding grazer respiration (ER); gross primary production (GPP) and net ecosystem production (NEP).

	Year 1 (2013/14)	Year 2 (2014/15)	Year 3 (2015/16)	3-Year Average	30-Year Normal
(a) Environmental variables					
PPT (mm)	987	1127	1104	1073	1250
T_A (°C)	13.6	13.5	13.9	13.7	13.3
$T_{A\ Jan\text{-}Mar}$ (°C)	17.2	18.2	19.3	18.3	17.7
VMC < PWP	18	47	0	22	n/a
(b) CO_2 Flux Components (g C m^{-2} y^{-1})					
ER	2387	2389	2800	2525	n/a
GPP	2751	2634	2965	2783	n/a
NEP	364 ± 26	245 ± 12	164 ± 9	258 ± 16	n/a

$m^3\ m^{-3}$; McLeod et al., 2016) of the soil (18 days in Year 1, and 47 days in Year 2; Fig. 3c). Regular summer rainfall in Year 3 resulted in the VMC getting close to, but not below, PWP.

Average annual air temperature for all three years was above the 30-year normal of 13.3 °C (NIWA, 2018). Annual air temperature for Years 1 and 2 (13.6 °C and 13.5 °C) was similar while Year 3 was warmer (13.9 °C) (Fig. 3b). The warmer annual temperature in Year 3 was mainly driven by temperatures in the January to March period, which was 1.7 °C warmer than the 30-year normal (Table 2).

3.2. CO_2 dynamics

Annual NEP, excluding grazer respiration, varied between 164 and 364 g C m^{-2} y^{-1}, being the highest in Year 1 and lowest in Year 3 (Table 2b) indicating a net uptake of CO_2 by the pasture in all years. ER and GPP were similar in Years 1 and 2, although ER slightly increased and GPP slightly decreased from Year 1 to Year 2 resulting in the reduction in NEP for Year 2. Both GPP and ER were much larger in Year 3 (2965 g C m^{-2} and 2800 g C m^{-2}, respectively) than either of the first two years.

Fig. 4 shows the cumulative NEP, GPP, and ER for each of the three years. In all years, the cumulative ER and GPP were similar until early-mid January. Deviation of cumulative GPP of Years 1 and 2 from Year 3 (Fig. 4c) occurred at the same time the volumetric soil moisture content dropped below PWP. In Year 3, the VMC never dropped below PWP (Fig. 3c) thus allowing the pasture to continue growing and photosynthesising (albeit at a slower rate). In Year 1 VMC dropped below PWP between 25 February 2014 and 14 March 2014, while in Year 2 VMC dropped below PWP one month earlier on 24 January 2015 remaining so until 11 March 2015. The impact of the extended period of VMC below PWP in Year 2 can be seen in the lower cumulative GPP relative to the other two years.

To allow flux measurements to integrate over a source area that included as many of the farm paddocks as possible, with the majority of the flux still originating from within the farm boundary, installation of the EC system was at a height of 2.72 m. The EC footprint (Fig. 1) extended over several of the paddocks surrounding the EC site, but the majority of the flux was derived from the paddocks immediately adjacent to the EC site. Contribution to the measured flux amounted to 40% for the paddock to the west of EC site, and 24% from the paddock to the east. One other paddock contributed 4%, while no other paddocks contributed more than 2.5%. As the majority of the flux contribution was derived from the two paddocks immediately adjacent to the EC system, their associated grazing regimes strongly influenced the measured CO_2 exchange (see Fig. 5b for an example period). Highest fluxes were observed in the day(s) immediately prior to grazing of the adjacent paddocks and correlated with biomass availability (data not shown). Following grazing, and the associated rapid removal of biomass, daytime fluxes were greatly reduced before slowly increasing

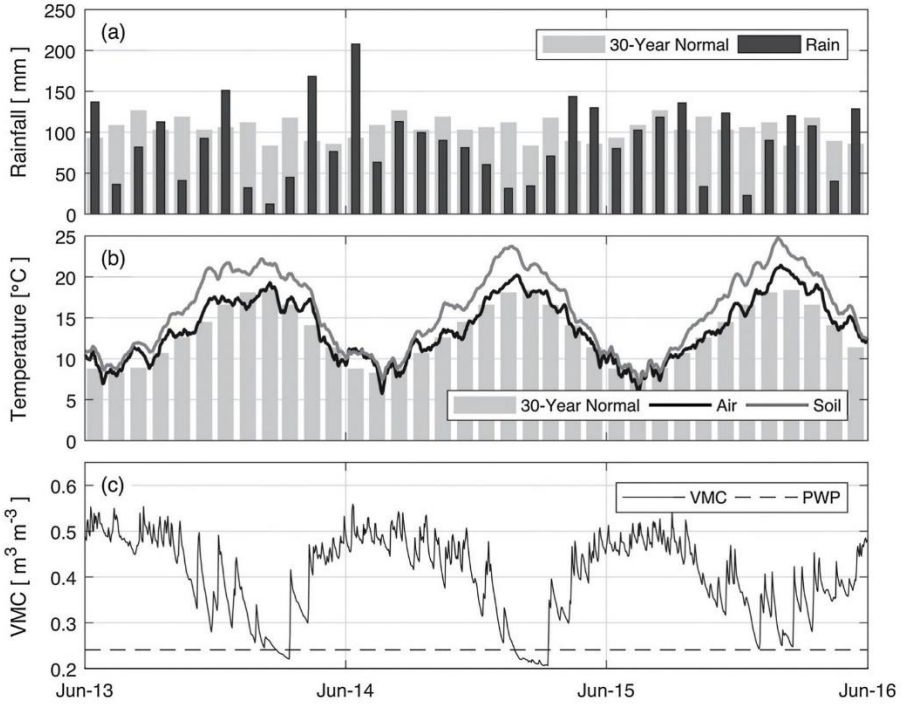

Fig. 3. (a) measured, and 30-year normal rainfall; (b) measured air temperature, 30-year normal air temperature, and measured soil temperature at 5 cm depth (air and soil temperature are 15-day running means); and (c) daily volumetric soil moisture content (VMC) at 10 cm depth, and permanent wilting point (PWP). Normal rainfall and air temperature were measured at a nearby weather station for the period.1981–2010.

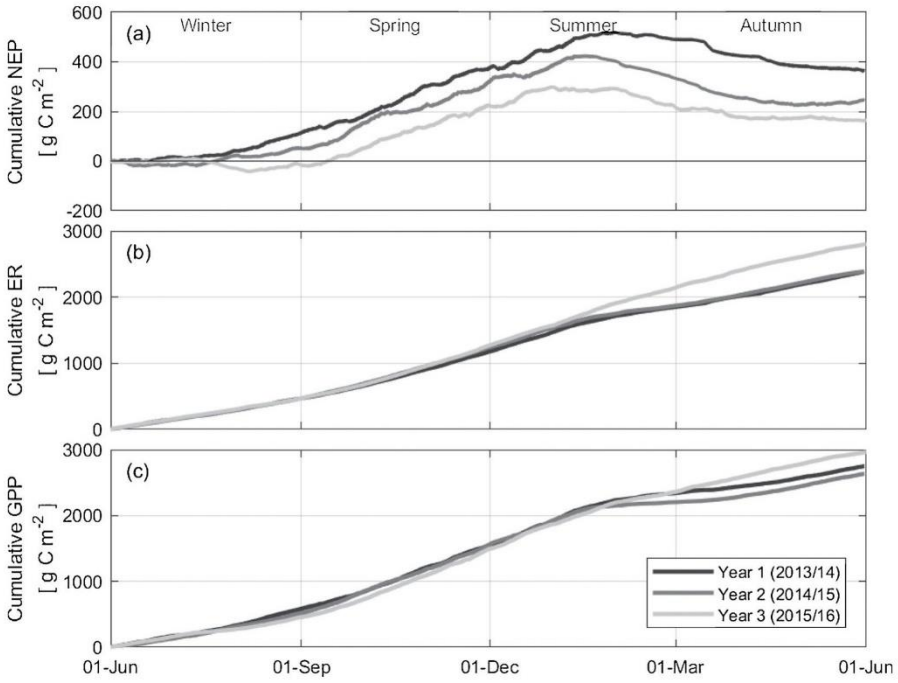

Fig. 4. Annual cumulative (a) NEP, (b) ER, and (c) GPP for each of the measurement years.

over the subsequent days until the next grazing event. The grazing impact of the remaining paddocks in the EC footprint was not easily observable in the flux data. Despite the EC footprint covering several paddocks, the annual NEP was dominated by the two paddocks adjacent to the EC system, and primarily controlled by their associated grazing regime.

All three years exhibited the same general pattern of strongest daytime CO_2 uptake in spring to early summer, with a period in summer of minimal CO_2 uptake (Fig. 5a) associated with the much lower soil moisture contents (Fig. 3c).

3.3. NECB

Carbon balances were calculated for $NECB_{Farm}$ and $NECB_{Footprint}$ with all components and the balance shown in Table 3. We consider $NECB_{Farm}$ to be the more accurate method, as we were more certain of the data quality of the non-NEP components for this system boundary. Firstly, $NECB_{Farm}$ and its components are reported, followed by $NECB_{Footprint}$ and its components.

3.3.1. $NECB_{Farm}$

For $NECB_{Farm}$, a positive C balance was obtained in all three years, although when uncertainties were considered only Year 1 resulted in a significant gain in C (131 ± 85 g C m^{-2} y^{-1}). Year 2 had the smallest C balance (35 ± 73 g C m^{-2} y^{-1}), while Year 3 was similar with a C balance of 45 ± 73 g C m^{-2} y^{-1}, and both years were considered to be C-neutral. The three-year average C balance was 71 ± 77 g C m^{-2} y^{-1}. Annual uncertainties were of similar magnitude to those found in other grazed pastoral systems using similar methodologies (e.g. Rutledge et al., 2015; Felber et al., 2016a), despite estimating (often conservatively) the uncertainties of the non-NEP components of the carbon balance.

Total C imports to the farm ranged from 715 g C m^{-2} y^{-1} (in Year 2) to 866 g C m^{-2} y^{-1} (in Year 1). Supplemental feed was the largest C import component, with the other major component being NEP. NEP was reduced in each of the years relative to the previous year, however, in Year 3, this decrease was compensated for by increased supplemental feed import resulting in higher net C imports in Year 3 than Year 2 (although still lower than Year 1). Total C exports were more constrained than the C imports, ranging from 680 to 750 g C m^{-2} y^{-1}. Carbon lost as animal respiration ($F_{Cow\ Resp}$; calculated from ingested pasture and supplemental feed) was the largest component to the total C lost from the system, accounting for approximately two-thirds of the total losses. Because grazer respiration was excluded from the NEP term, any CO_2 respired by the grazing animals while within the EC footprint was included in the respiration term rather than NEP. It should be noted that our method to estimate grazer respiration resulted in a lower percentage of intake being respired than other studies (see supplementary material Table S3 for data), however, the daily per cow respiration rates were similar to other measured values for dairy cattle (e.g. Kinsman et al., 1995). Milk production was the second major pathway for C loss, accounting for 27% of the total C export. Milk export in Year 2 was larger than Year 1 despite having lower supplemental feed import and pasture production (Table 1). Approximately half of the difference in milk export between Years 1 and 2 can be attributed to a larger herd size (15 cows larger in Year 2 (Table 1)) with the other half of the difference occurring during the August to October period, where pasture growth was greater than Year 1 providing the cows with more high-quality feed. Therefore, despite overall total C ingested (in the pasture and supplemental feed) being lower, the timing and availability of different feed types may have affected the final C balance.

3.3.2. $NECB_{Footrpint}$

The three-year average $NECB_{Footprint}$ was 56 ± 77 g C m^{-2} y^{-1}, or 15 g C m^{-2} y^{-1} less than the three-year average $NECB_{Farm}$ (71 ± 77 g C m^{-2} y^{-1}). In general, the $NECB_{Farm}$ and $NECB_{Footprint}$ calculations gave similar annual C balances, with the differences being 16, 16 and 14 g C m^{-2} y^{-1}, respectively, for each of the three years (Table 3). In all years, $NECB_{Farm}$ was greater than $NECB_{Footprint}$. NEP, fertiliser, and

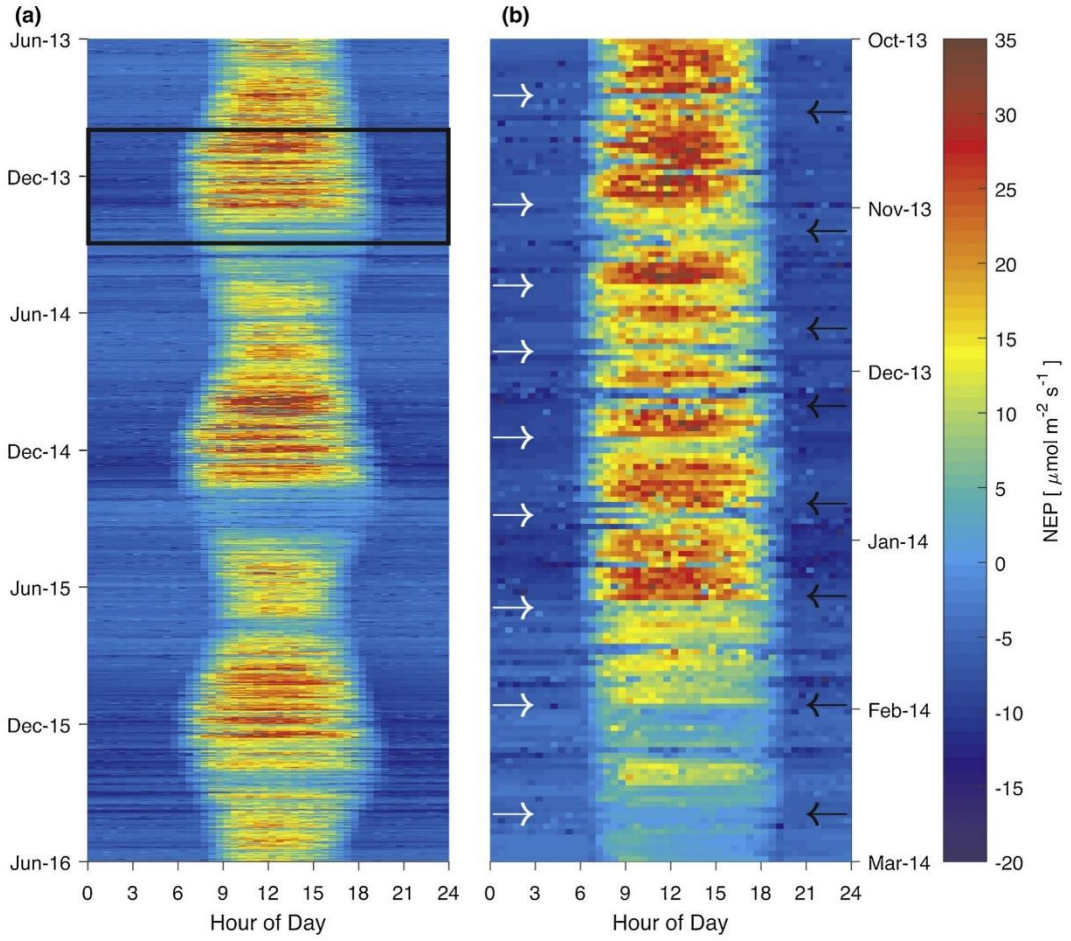

Fig. 5. Half-hourly gap-filled NEP for the entire measurement period (1 June 2013 to 31 May 2016) are shown in (a), and a subset (1 October 2013 to 1 March 2014) illustrating the impact of grazing in paddocks adjacent to the eddy covariance site are shown in (b). The black box in (a) surrounds the period shown in (b). The arrows in (b) identify grazing events in the paddocks west (white arrows, left side of plot) and east (black arrows, right side of plot) of the EC site.

leaching components were identical in both methods, and while the supplement feed imported component was also in both equations, the magnitude was different depending on the system boundary location. Imported supplement feed contributing to $NECB_{Footprint}$ only included that fed directly onto paddocks within the EC footprint (grass silage and maize husk), thus excluding all maize silage and PKE imports. Consequently, imported supplement feed entering the $NECB_{Footprint}$ system boundary averaged 25 g C m^{-2} y^{-1} compared to 526 g C m^{-2} y^{-1} entering the $NECB_{Farm}$ system boundary. Total C imports for $NECB_{Footprint}$ decreased in each of the three years, primarily driven by the decrease in NEP, which was either the largest (Year 1) or second largest import (Years 2 and 3). Excreta deposition was the other major import, averaging 318 g C m^{-2} y^{-1}, and like effluent application, was strongly influenced by the quantity and quality of the supplemental feed ingested by the cows on the feed pad. Carbon exported from the $NECB_{Footprint}$ system boundary was almost exclusively a function of the removal of biomass, either as grazed pasture ($F_{Grazing}$) or mechanically harvested pasture ($F_{Harvest}$). Like C imports, C exports decreased in each of the three years, correlating with lower annual pasture growth (Table 1).

4. Discussion

4.1. Comparison of predicted and measured annual NECB

Following importation of supplemental feed of between 10.4 and 13.5 t DM ha^{-1} y^{-1} (461–622 g C m^{-2} y^{-1}) the average annual C balance for the rotationally grazed dairy farm was 71 ± 77 g C m^{-2} y^{-1}. As the uncertainty bounds pass through zero, we were unable to fully conclude whether or not the Ryan farm was gaining soil C, however, it was unlikely to have lost soil C. Perhaps our results are not unexpected as other approaches also suggested the imported supplemental feed would lead to a relatively small gain in soil C, the magnitude of which was much smaller than the size of our uncertainties. For example, estimates of theoretical gains in soil C resulting from the imported supplemental feed can be made using modelling (Kirschbaum et al., 2017), and stabilisation of C from additional excreta (Maillard and Angers, 2014). Kirschbaum et al. (2017) modelled the change in soil C and milk production as a function of increased supplemental feed import, reporting gains in both milk production and soil C that scaled approximately linearly with the quantity of additional supplemental feed. Extrapolating their modelled results for the Ryan Farm (where the average supplemental feed import was 11.6 t DM ha^{-1} y^{-1}, or 526 g C

Table 3
NECB components calculated as per Eq. (1) for $NECB_{Farm}$, and (2) for $NECB_{Footprint}$. Year 1 was 1 June 2013–31 May 2014; Year 2 was 1 June 2014–31 May 2015; and Year 3 was 1 June 2015–31 May 2016. All units are g C m^{-2} y^{-1}, and n/a indicates component is not applicable for that system boundary. The uncertainties for each term are reported as standard deviations, while the uncertainty for the average column was calculated as the arithmetic mean of the three years.

	$NECB_{Farm}$				$NECB_{Footprint}$			
Component	Year 1	Year 2	Year 3	*Average*	Year 1	Year 2	Year 3	*Average*
NEP (F_{CO_2})	364 ± 26	245 ± 12	164 ± 9	*258 ± 16*	364 ± 26	245 ± 12	164 ± 10	*258 ± 16*
Synthetic fertiliser ($F_{Syn\ Fert}$)	9 ± 2	9 ± 2	9 ± 2	*9 ± 4*	9 ± 2	9 ± 2	9 ± 2	*9 ± 2*
Excreta ($F_{Excreta}$)	n/a	n/a	n/a	n/a	337 ± 27	294 ± 24	324 ± 39	*318 ± 30*
Effluent ($F_{Effluent}$)	n/a	n/a	n/a	n/a	49 ± 4	43 ± 3	51 ± 4	*48 ± 3*
Supplement feed ($F_{Sup\ Feed}$)	494 ± 20	461 ± 18	622 ± 25	*526 ± 53*	25 ± 4	41 ± 5	9 ± 1	*25 ± 3*
Supplement feed removed ($F_{Sup\ Feed\ Removed}$)	n/a	n/a	n/a	n/a	–20 ± 3	–33 ± 3	–7 ± 1	*–20 ± 2*
Pasture removed by grazers ($F_{Grazing}$)	n/a	n/a	n/a	n/a	–620 ± 76	–536 ± 67	–518 ± 61	*–558 ± 68*
Respiration ($F_{Cow\ Resp}$)	–499 ± 77	–434 ± 69	–491 ± 67	*–475 ± 47*	n/a	n/a	n/a	n/a
Methane (F_{CH4})	–37 ± 6	–32 ± 5	–36 ± 5	*–35 ± 3*	n/a	n/a	n/a	n/a
Milk (F_{Milk})	–180 ± 9	–195 ± 10	–201 ± 10	*–192 ± 10*	n/a	n/a	n/a	n/a
Leaching ($F_{Leaching}$)	–1 ± 0	–2 ± 2	–1 ± 1	*–1 ± 1*	–1 ± 0	–2 ± 2	–1 ± 1	*–1 ± 1*
Pond emissions ($F_{Pond\ Emissions}$)	–19 ± 3	–17 ± 2	–21 ± 3	*–19 ± 4*	n/a	n/a	n/a	n/a
Harvest ($F_{Harvest}$)	n/a	n/a	n/a	n/a	–28 ± 3	–42 ± 5	0	*–23 ± 3*
NECB	131 ± 85	35 ± 73	45 ± 73	*71 ± 77*	115 ± 85	19 ± 73	31 ± 74	*56 ± 77*

m^{-2} y^{-1}) would predict a gain in soil C of 24 g C m^{-2} y^{-1} over a 50-year period. However, Kirschbaum et al. (2017) noted that gains are not likely to be linear through time, with more rapid rates of storage in the first few years. The importation of large quantities of supplemental feed at the Ryan Farm had only begun 6 years prior to this study (W. Ryan pers comm, 2013), and thus the modelling would suggest a higher sequestration rate during our measurement years (of ~40 g C m^{-2} yr^{-1}) than would have been expected over the 50-year period.

While importing supplemental feed introduced more C to the dairy farm system, consumption of this feed by the grazing animals resulted in the majority of the additional C being lost shortly after ingestion as animal respiration, methane, and milk. Therefore, only C excreted by the grazing animals, and any additional C wasted during the feed storage, supply and consumption was available for sequestration (Fig. 7). Good management of the feed storage, supply and consumption processes results in minimal wastage (< 5%; DairyNZ, 2017), with the farm manager confirming this was the case for the duration of this study (W. Ryan pers comm. 2014). Therefore, the supplemental feed C actually available for sequestration was predominantly through the direct deposition of excreta, and application of effluent onto the soil. Following a meta-analysis, Maillard and Angers (2014) report a global manure-C retention coefficient of 12%, i.e. the proportion of manure stored as soil C. Applying this coefficient to the supplemental feed generated excreta applied to the soil (direct deposition plus effluent applied) indicated an expected C gain of 25, 22 and 29 g C m^{-2} y^{-1}, respectively, for the three measurement years.

Both the modelling and meta-analysis studies indicate only small gains in soil C could be achieved by the importation of supplemental feed regardless of the quantity imported. Consequently, detection of an increase in soil C attributable to supplemental feed would be difficult to measure over annual time scales using the NECB method due to the magnitude of the uncertainties relative to the size of the expected gain. Over several years the cumulative gains may be larger and therefore detectable using other methodology (i.e. temporal soil resampling). Additionally, our measured C balance is likely also further confounded by non-management related effects (e.g. climatic) that could result in short term C loss or gain. Even if a control was imposed to account for external factors, detecting such small differences would be challenging as replicating the exact management without the imported feed would be impossible. Without the imported supplemental feed, insufficient feed would be available to maintain an identical (high) stocking rate, and thus additional management changes would also occur. Any aligned control experimental has the potential to introduce spatial variability adding further complication, and thus further limiting the ability to detect the expected small change. Our study demonstrated that importing a large quantity of supplemental feed C (up to 50% of total diet and between 10.4–13.5 t DM ha^{-1} y^{-1}) did not lead to a large gain in soil C. Furthermore, any potential gains would likely be offset by GHG emissions during production of the imported feed (see below).

4.2. Impact of supplemental feed on CO_2 dynamics

One of the key components of agricultural carbon balance studies is the exchange of CO_2 between the ecosystem and atmosphere (i.e. NEP), and indeed several studies focus purely on this component (e.g. Jérôme et al., 2014; Gourlez de la Motte et al., 2019). Therefore, the impact supplementary feed has on NEP and its component terms, ER and GPP, also needs to be considered.

4.2.1. GPP and ER

Ryan Farm GPP and ER (three-year averages of 2783 g C m^{-2} yr^{-1} and 2525 g C m^{-2} yr^{-1}, respectively) were greater (12–32%) than those determined from three previous studies in the Waikato region of New Zealand (Campbell et al., 2015; Rutledge et al., 2015, 2017b). All three comparative studies had stocking rates close to the regional average stocking rate (2.95 cows ha^{-1}; DairyNZ, 2016) indicating those farm systems operated at a lower intensity than the Ryan Farm. The higher GPP and ER observed from the more intensively managed Ryan

Farm was in agreement with findings of Allard et al. (2007). The use of supplemental feed to allow Ryan Farm to operate at a greater intensity and increased the amount of excreta deposited onto the pasture, with increased excreta having been shown to lead to an increase in total respiration (Shimizu et al., 2009). Assuming a 12% C retention coefficient (Maillard and Angers, 2014), 88% of the excreta would be eventually respired corresponding to the ER of the Ryan Farm being approximately 155 g C m^{-2} y^{-1} greater than if no supplemental feed was used. However, supplemental feed is likely to have less impact on GPP. Shimizu et al. (2015) report GPP for a grassland site in Hokkaido, Japan, with two treatments; (i) addition of synthetic fertiliser only, and (ii) addition of synthetic fertiliser and manure. They found that there was no difference in GPP between the two treatments, therefore, suggesting the addition of manure resulted in no change in GPP. Combining the effect of additional excreta on GPP (no change) with to that of ER (increasing ER) suggests increased supplemental feed alone would lead to a reduction in NEP. As Ryan Farm also utilised considerable N fertiliser, which likely increases GPP (Allard et al., 2007; Shimizu et al., 2015), the resultant GPP and ER, and thus NEP of this study, was the combined effect of the overall management system.

4.2.2. NEP

While the impact supplemental feed has on NEP is controlled by its effect on GPP and ER (as described in the previous section), it is worthwhile briefly considering what the reported NEP represents. As Gourlez de la Motte et al. (2019) highlight, there is considerable inconsistency in how NEP (or NEE in other studies) is reported in grassland studies, particularly as inclusion of grazer respiration in EC CO_2 flux measurements have been identified as problematic (Skinner, 2008; Kirschbaum et al., 2015). Some studies (e.g. Rutledge et al., 2015; Jones et al., 2017) include grazer respiration in NEP, while others do not (e.g. Hunt et al., 2016; Rutledge et al., 2017a), with some more recent studies reporting both (Felber et al., 2016b; e.g. Gourlez de la Motte et al., 2019). In this study, we excluded grazer respiration from NEP. The three measurement years reported a CO_2 sink of between 164–364 g C m^{-2} y^{-1}, which was similar to other comparable grassland studies where grazer respiration was excluded (Felber et al., 2016b; Hunt et al., 2016; Gourlez de la Motte et al., 2019). Total average grazer respiration for the Ryan Farm of 475 g C m^{-2} y^{-1} (Table 3) was considerably higher than that reported in other studies (e.g. 189 g C m^{-2} y^{-1} in Jérôme et al. (2014) and 180 g C m^{-2} y^{-1} in Felber et al. (2016b)). Addition of grazer respiration to NEP allows calculation of total NEP for the ecosystem (NEP_{tot}), which for the Ryan Farm would average −216 g C m^{-2} y^{-1} (i.e. a loss of C). Such a result would be one of the lowest reported NEP values from any grassland study. Because animal respiration is dependent on the total intake of feed, there were contributions from both on-farm produced pasture (57% of diet) and imported supplemental feed (43% of diet). While it would be impossible to directly estimate NEP_{tot} had there been no imported supplemental feed (due to the uncertainty around how supplemental feed impacts GPP and ER), it is worthwhile to consider that the imported supplemental feed contributed 46% to total grazer respiration (or 218 g C m^{-2} y^{-1}). To summarise, the direct impact that supplemental feed has on NEP excluding grazer respiration is difficult to quantify, however, the addition of grazer respiration to NEP would result in a negative NEP_{tot} for this site largely due to grazer respiration attributable to the imported supplemental feed.

4.3. Production of supplemental feed effect on soil C and GHG emissions

It is important to consider the effect of production of the imported supplemental feed on soil C. The main supplement feed types used by the Ryan Farm were locally produced maize silage and internationally imported PKE. No New Zealand studies have investigated the NECB of the production of maize silage. Where soil properties of pasture sites have been compared to nearby maize cropping sites, soil C stocks were found to be considerably lower under maize cropping (Sparling et al., 1992; Shepherd et al., 2001; Stevenson et al., 2016), with the rate of loss dependent on the cropping duration (Shepherd et al., 2001). Internationally, NECB studies of maize production have also demonstrated soil C losses (Zeri et al., 2011; Wang et al., 2015; Eichelmann et al., 2016). Quantifying the impact of PKE production on soil C is more challenging, but important because of its use in New Zealand farm systems following importation from South East Asia. Indeed, New Zealand farms which use considerable quantities of PKE have been shown to markedly increase their carbon footprint relative to those that do not (Reisinger et al., 2017). Because the production and usage of supplemental feed utilises much mechanisation to produce, harvest, transport, then (repeatedly) provide the feed to the grazing animals, a total picture of the true impact of supplemental feed usage would only emerge following a complete life cycle assessment. Such as an assessment would also need to include the contribution of other greenhouse gases (e.g. N_2O and CH_4).

We showed that importing a large quantity of supplemental feed did not lead to a large gain in soil C, but expected losses of soil C during feed production suggests the combined processes likely leads to an overall large loss of soil C.

4.4. Methodology considerations for grazed pastoral systems

We compared the C balance calculated by two methods ($NECB_{farm}$ and $NECB_{footprint}$) differing by their respective system boundaries. For all measurement years the difference between the calculated NECB's was small (< 16 g C m^{-2} y^{-1}), with $NECB_{Farm}$ always higher than $NECB_{Footprint}$. Each method required differing terms in the C balance calculation and thus highlighted different components as being important. It is worthwhile noting that in this study the two NECB calculation methods were not truly independent checks on each other, but rather intended to highlight that similar NECB's can be obtained using different system boundaries, which can impact interpretation of driving factors. For example, in the calculation of $NECB_{Farm}$ supplemental feed import and animal respiration were the major terms (Table 3; Fig. 6). In contrast, with the system boundary located around the EC footprint ($NECB_{Footprint}$), direct supplemental feed imports were only a minor component of the NECB despite being responsible for 43% of the total diet during the three years, and animal respiration was not included. Consumption of the supplemental feed outside the $NECB_{Footprint}$ system boundary was indirectly included in the C balance calculation as excreta transfer. Consequently, interpretation of the C balance components to identify processes (management or otherwise) influencing the overall C balance needs to be carefully considered, as the magnitude of any term may be simply a function of the location of the system boundary.

Choice of which system boundary is suitable depends on available data. While calculating the NECB using two system boundaries for the Ryan Farm, we considered $NECB_{Farm}$ as the most accurate because most of the non-NEP data available were provided for the farm in its entirety. For example, supplemental feed import and milk export information was provided as the quantity that passed through the farm gate, while pasture growth was provided as an average of the farm. While this information can be manipulated to allow calculation of $NECB_{Footprint}$, as done here, it required the assumption that the data for the farm is applicable to the area within the EC footprint. As our EC footprint extended over approximately one-third of the farm, such an assumption would be more valid than if the EC footprint occupied only a couple of paddocks. Regardless of the system boundary location, key measurements in grazed pasture systems are those of the animal C cycle. In the calculation of $NECB_{Footprint}$, where the grazing cows were considered agents of export, only the C removed by grazing and the excreta returned were required to be quantified. However, for $NECB_{Farm}$, closure of the animal C cycle was required, as losses from the system included milk, cow respiration and methane (deposited excreta remained within

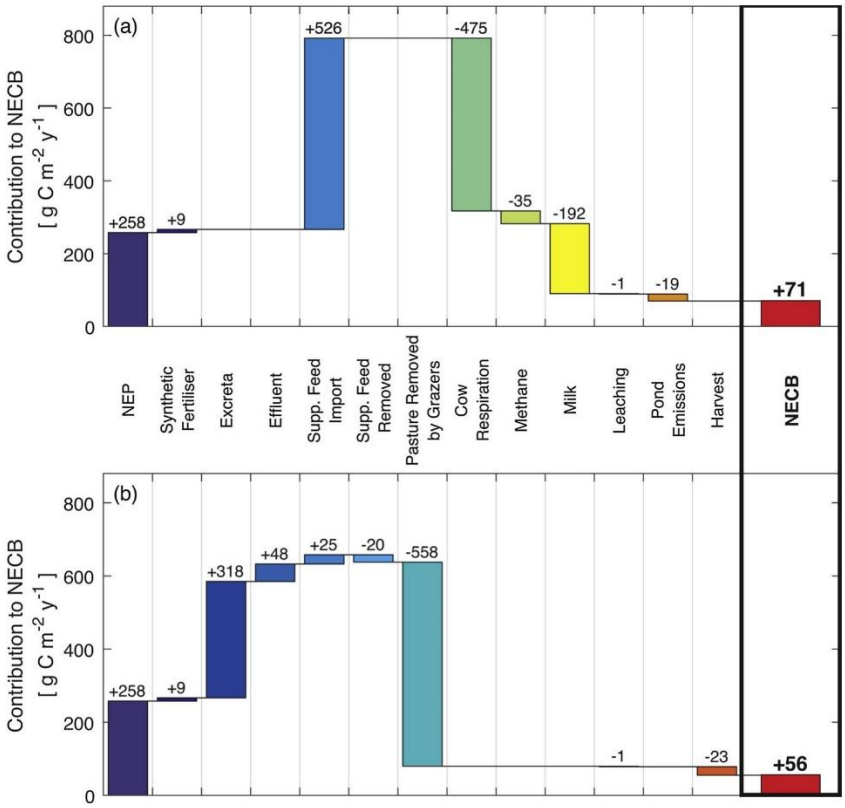

Fig. 6. 3-year mean NECB and components at both the $NECB_{Farm}$ (a) and $NECB_{Footprint}$ (b) system boundaries. Each bar shows the magnitude of the component and is cumulative from the previous bar. Values at the top of the bar represent the magnitude and direction of the component (positive values are a gain of C, and negative values a loss of C). The NECB is the sum of all components and is displayed as the furthest right bar inside the black box.

the system boundary). Our approach was to quantify the intake, excreta deposition and milk export, then close the balance to estimate the respiration and methane components. This approach, while using more accurate data (through known milk export, digestibility and C intake), resulted in proportionally larger uncertainties on $F_{Cow\ Respiration}$ and $F_{Methane}$ due to requiring the propagation of more terms to calculate each component. Conversely, uncertainty estimates for the $F_{Grazing}$ and $F_{Excreta}$ components of $NECB_{Footprint}$ were smaller due to the propagation of fewer (less accurate) terms in their calculation. The result was almost identical NECB uncertainties from both methods. $F_{grazing}$ could have been further partitioned into the animal C component terms (rather than simply reporting $F_{grazing}$), however, the result would have increased uncertainty on each term through the propagation of the uncertainties. In choosing a system boundary, consideration must be given not only to what data is available (or to be measured) but also to how the uncertainty is to be calculated and thus enabling the calculation of the most accurate NECB coupled with a minimised uncertainty.

In this study, choosing a system boundary bordering the farm required the key assumption that the measured NEP was representative of the entire farm. Management of the farm was extremely regimented whereby the total herd was divided into three herds of equal size, each receiving the same feed quantity daily, and grazing the same area of pasture each day. Paddock sizes were similar, while the topography was flat, and soils mostly uniform across the farm. Thus, we concluded that the measured NEP was representative of the farm. However, during the measurement period, two paddocks on the farm were subjected to an additional management practise – pasture renewal. The two paddocks that underwent pasture renewal were outside the EC footprint and therefore effects of this process (on either NEP or NECB) were not captured in $NECB_{Farm}$. Quantification of the effect of pasture renewal on soil C stocks at an annual basis is somewhat unclear (Ammann et al., 2013; Rutledge et al., 2015, 2017b) and therefore we chose to ignore this component. However, had we incorporated a loss of say, 200 g C m^{-2} y^{-1} for the two renewed paddocks, the net effect on $NECB_{Farm}$ would be a decrease of ~10 g C m^{-2} y^{-1} for Year 2, and ~3 g C m^{-2} y^{-1} to the three year average, hence having little effect on our findings.

5. Conclusions

The net ecosystem carbon balance (NECB) for a rotationally grazed dairy farm importing 11.6 t DM ha^{-1} y^{-1} as supplemental feed (equivalent to 43% of the grazers' diet) averaged 71 ± 77 g C m^{-2} y^{-1} for three years. The magnitude of this result was similar to that predicted from system modelling and calculations of C sequestration from additional excreta due to supplementary feed. Our measured NECB, coupled with an understanding that most of the imported supplementary feed C was removed from the system following animal consumption, allow us to conclude that importing large quantities of supplemental feed C did not correspond to a large increase in soil C on an annual basis. Accumulation of smaller annual gains resulting from the imported supplemental feed may be occurring, but different methodology would be required to determine if this is the case. Not considered here, but worthy of future exploration is what effect production and export of the supplemental feed has on soil C at the location of the production, and the net change in soil C is when considering both production and usage of this feed. Expansion of such a concept could also include a full life cycle assessment including fossil fuel consumption to enable the production and usage of supplemental feed.

We also explored the impact of the location of the system boundary on NECB, calculating the C balance with a system boundary bordering the farm ($NECB_{Farm}$) and including only the paddocks within the EC footprint ($NECB_{Footprint}$). Both the calculated C balances and

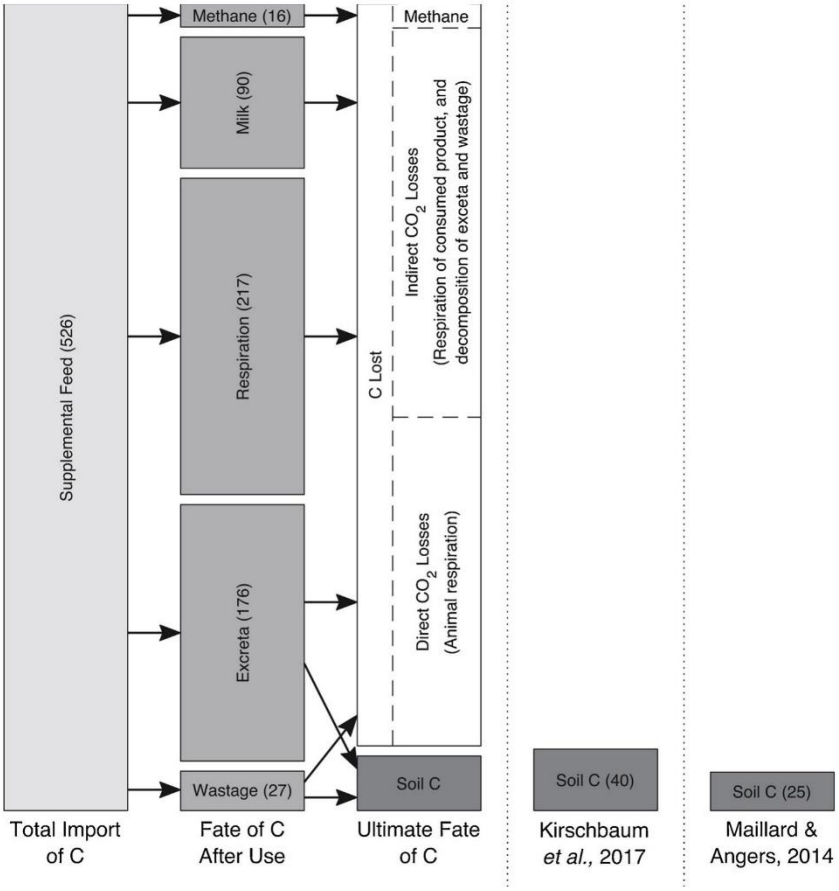

Fig. 7. Conceptual diagram of the flows and fate of imported supplemental feed C. Where applicable, the values in brackets after each component represent the magnitude of the component as calculated in this study (in units of g C m^{-2} y^{-1}), with the height of all bars proportional to these values. The arrows indicate the transformation of C through time. For comparison, predicted gains in soil C for the quantity of supplemental feed imported based on literature information are also displayed.

uncertainties were very similar for both approaches in all years. Choice of the system boundary location can be somewhat dictated by the quality and availability of data, however, consideration must also be given to how uncertainty is to be calculated, with the dual aim of increasing NECB calculation accuracy coupled with minimising uncertainty. Additionally, any assumptions, particularly with respect to the area that the measured NEP component represents, must be carefully considered. Finally, different system boundary locations can result in variation to the NECB components and their magnitude, potentially impacting interpretation of factors influencing the C balance.

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Appendix A. Supplementary data

Supplementary material related to this article can be found, in the online version, at doi:https://doi.org/10.1016/j.agee.2019.04.019.

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Chapter 4:

Quantifying carbon losses from periodic maize silage cropping of permanent temperate pastures

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Please refer to Appendix D for the contribution of each of the authors

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Quantifying carbon losses from periodic maize silage cropping of permanent temperate pastures

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ABSTRACT

New Zealand dairy farming systems use considerable maize silage as a supplemental feed, often producing the crop within the farm boundary by cropping a paddock for one or more years then rotating through other areas of the farm. While globally the effects of permanent maize (*Zea mays* L.) cropping on soil carbon (C) stocks have been studied, little is known about the impact short-duration periodic cropping for maize silage within rotationally grazed pastures has on soil C stocks. In this study, we used the net ecosystem carbon balance (NECB) approach to determine the carbon budget of two maize silage crops in subsequent years and contrasted this with the alternative of grazed pasture on the same farm. Eddy covariance measurements provided CO_2 exchange data, which were coupled with measurements or estimates of all other carbon imports and exports of the two systems. We chose to isolate just the period of maize silage production with our measurements extending from the time of herbicide application before planting of the maize crop through to seedling emergence of pasture re-establishment following harvest. The NECB of the maize silage system identified a large loss of C from the ecosystem (–850 g C m^{-2} and –415 g C m^{-2} for years 1 and 2 respectively), while the pasture was near neutral in year 1 (11 g C m^{-2}) and a C source in year 2 (–114 g C m^{-2}). Aboveground biomass production from the maize system was three times greater than the pasture system, with > 90 % harvested and removed. In contrast, a net of ~60 % of the aboveground biomass was removed from the pasture system. The large difference in exported biomass, coupled to long establishment periods with no photosynthetic C inputs of the maize were key factors in the much larger C losses from the maize system relative to the pasture.

1. Introduction

Global agricultural ecosystems are both major sources of greenhouse gas emissions (GHG; Smith et al., 2014), yet through the soils on which the agricultural activities occur has the potential to mitigate emissions through carbon (C) sequestration (Minasny et al., 2017). Recently, there has been a focus on improving the understanding of these systems to meet dual goals of improving productivity and identifying management options that maximise the mitigation potential. Accordingly, the C dynamics of agricultural systems such as croplands and grazed grasslands have been increasingly studied, but often independently. Much less consideration has been given to coupled systems where grazed pastures are intermittently cropped to produce feed for the grazing animals.

New Zealand dairy systems utilise arable crops to provide supplementary feed for grazing cattle. Increasingly more supplementary feed is being used both to provide feed during times of low pasture growth (Fausett et al., 2015), and to increase milk production (Densley et al., 2001). Many non-pasture supplements are used within the dairy industry, with the largest locally produced being maize silage (DairyNZ Economics Group, 2016), with the cropping area increasing by 144 % between 2002 and 2017 (Statistics New Zealand, 2004, 2018). Maize for silage is produced in two distinct ways: (1) grown on permanent croplands (cropping farms or dairy support blocks) and exported to the dairy farm where it is fed (64 %; Booker, 2009); or (2) grown on the farm where it will be fed (36 %; Booker, 2009). The first of these options often leads to crops repeatedly produced on the same land, while the land area used to grow the crop in the second option is frequently rotated around the farm. Maize silage produced on-farm generally forms part of the farmers' pasture renewal process where exhausted

Abbreviations: C, carbon; EC, Eddy covariance; ER, ecosystem respiration; GHG, greenhouse gas; GPP, gross photosynthetic production; NECB, net ecosystem carbon balance; NEE, net ecosystem exchange; NEP, net ecosystem production; PWP, permanent wilting point; VMC, volumetric moisture content

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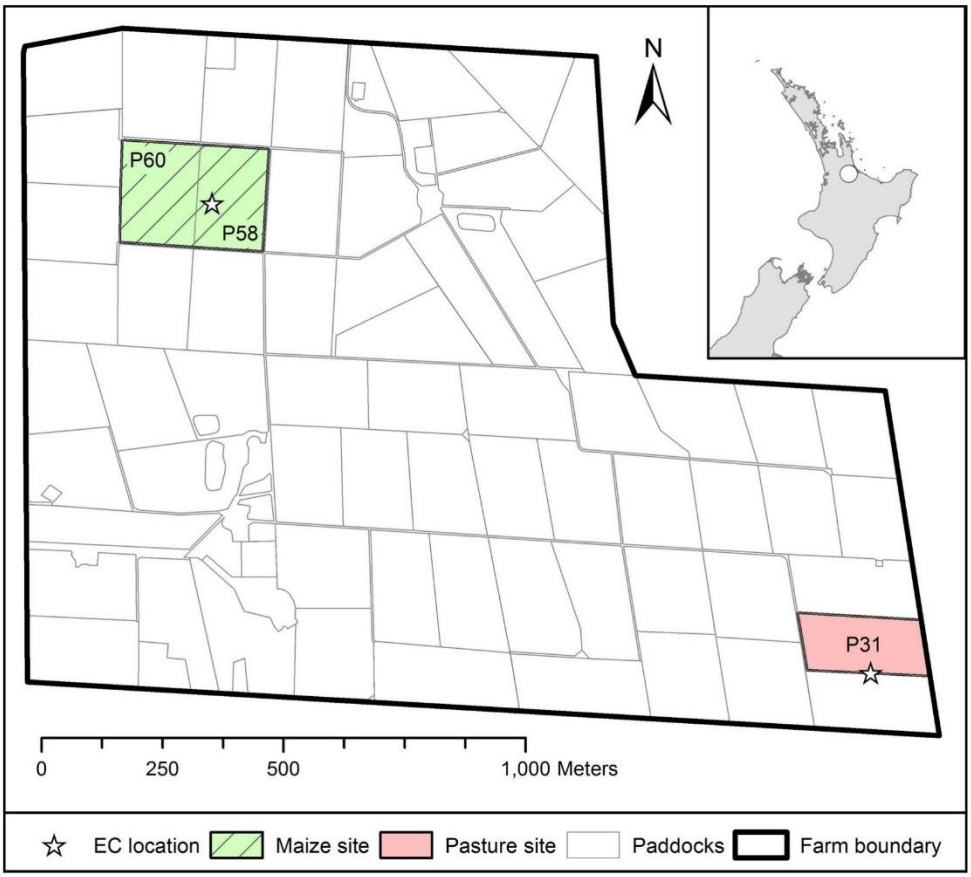

Fig. 1. Map of Troughton Farm identifying the maize and pasture sites, and their respective eddy covariance locations. Inset map locates Troughton Farm within the North Island of New Zealand.

pastures are first cropped for maize silage before being returned to permanent pasture (Densley et al., 2011). In such circumstances, the maize cropping may only be for one year before returning to a permanent pasture sward for a decade or more, and therefore represents periodic rather than continuous cropping.

Previous studies reporting the carbon dynamics from cropping systems that include maize (or corn) often identified those sites to be a source of C (up to 7–8 t C ha^{-1} y^{-1}; Poyda et al., 2019) that is generally larger than other crop types (Ceschia et al., 2010; Poyda et al., 2019). These studies are mostly from long-term continuous cropping systems and rarely focus on the impact of the maize crop alone. The maize crop may be coupled to a winter crop (Bao et al., 2014; Wang et al., 2015) or a fallow period (Zeri et al., 2011; Eichelmann et al., 2016) with the resultant carbon balances (assumed to represent the change in soil C) reported on an annual basis and, therefore, do not explicitly quantify the maize effect alone. The carbon balances are further complicated by the purpose of the maize crop; for example, production for maize silage and some biofuels remove most biomass C (e.g. Zeri et al., 2011; Lai et al., 2017), while grain only harvests may leave considerable stover (stalks and leaves) residue for future incorporation into the soil (e.g. Wang et al., 2015). Furthermore, changes to soil C stocks within cropping systems (maize or otherwise) can be achieved by manipulating management processes such as tillage intensity (Busari et al., 2015; Haddaway et al., 2017) and frequency (Conant et al., 2007; Soussana et al., 2010), nutrient supply through organic and synthetic fertilisers (Triberti et al., 2008; Hijbeek et al., 2016; Chen et al., 2018), and irrigation (Verma et al., 2005; Suyker and Verma, 2012; Wang et al., 2015). Consequently, changes in soil C for any maize cropping system are a function of crop purpose, management, site history and climate (Godde et al., 2016).

Although continuous long-term cropping systems have been extensively studied, there are very few studies of periodic cropping of grasslands on soil C stocks (Linsler et al., 2013). While long-term conversion from grasslands to permanent croplands results in a decline of the soil C stock (Guo and Gifford, 2002), there is evidence to suggest soil C lost during an occasional cropping event may be recovered within a few years. For example, following a single cropping event Linsler et al. (2013) found significant differences in soil C stocks observed after two years were no longer significant after five years. Densley et al. (2001) proposed periodic maize silage cropping followed by re-establishment of pasture of each area of a farm approximately every seven years as a measure to improve pasture growth and increase animal feed availability, although the authors did not consider impacts on soil C. The viability of such a system to maintain soil C stocks depends on the magnitude of the short-term C loss associated with the maize silage cropping process and whether the return period is sufficient for the loss to be regained. Estimation of the magnitude of C loss associated with occasional maize silage cropping is difficult to determine from existing research due to (1) the individuality of each system, and (2) most studies occurring on permanent croplands. For example, Wang et al. (2015) were one of the few studies to separate the maize cropping period from the winter period, however, their study included removal

of maize grains only, and measurements were from long-term continuously cropped land. There is evidence to suggest that soils with higher C contents (i.e. grasslands relative to croplands) have an increased susceptibility to losses of soil C (Kutsch et al., 2010; Godde et al., 2016), and therefore results from continuous cropping systems may underestimate losses from periodically cropped systems.

The objective of our study was to determine the carbon balance of two years of maize silage cropping following recent conversion from pasture and to compare these balances with that of rotationally grazed pasture. We hypothesised that maize silage production, as part of New Zealand dairy farming operations, would result in rapid initial losses of C through a management combination that included conventional tillage, full biomass removal, as well as short-term land use change (conversion of pasture to cropland). Carbon balances of both the pasture and maize sites were determined using the net ecosystem carbon balance technique (Chapin et al., 2006) for two cropping seasons prior to the site being returned to permanent pasture. We combined eddy covariance measurements of CO_2 exchange with measurements and estimates of all other flows of C into and out of each system. To isolate the effect of the maize crop, our measurement period extended from the start of the maize crop preparation and extended through to the date of seedling emergence of the replacement sward (i.e. returning the cropped site to either annual or permanent pasture) for each year.

2. Methods

2.1. Study site

2.1.1. Site description

Measurement sites for both the maize crop and pasture were located on Troughton Farm, a commercial dairy farm in the Waikato region of New Zealand (37° 45.6′ S, 175° 47.2′ E, 54 m.a.s.l). The maize site included two paddocks (P58 and P60) totalling 6.37 ha in area, while a single paddock (P31; 2.97 ha) was chosen for the pasture comparison (Fig. 1). An eddy covariance (EC) system was established in P58 of the maize site with the flux footprint including P60. The pasture site utilised an EC system located on the boundary of P31 (Rutledge et al., 2017a, b; Wall et al., 2020). Measurements were made for two maize crop seasons during the 2016/17 (year 1) and 2017/18 (year 2) summers. Previously, the paddocks had been under continuous permanent pasture composed of ryegrass (*Lolium perenne*) and white clover (*Trifolium repens*) and had been so for several decades. Like the maize site, the pasture site had been under permanent pasture (predominantly ryegrass and clover) for several decades and was last renewed in 2013 (Rutledge et al., 2017b).

Troughton farm is situated in a flat to gently undulating landscape with a complex of four soils (Waihou, Pairere, Te Puninga and Waitoa series of soils) formed from rhyolitic and andesitic volcanic ash on rhyolitic alluvium (McLeod, 1992). The maize crop site was located on the well-drained Waihou silt loam, which was classified as a Typic Orthic Allophanic Soil (Hewitt, 1998). The measured bulk density of the topsoil was 748 kg m^{-3}, while the carbon content of the A horizon was 8.03 %, both similar to typical values for the soil type (McLeod, 1992; McLeod et al., 2016), while the permanent wilting point was 0.24 m^3 m^{-3} (McLeod et al., 2016). Total carbon stocks to 0.6 m were measured between years 1 and 2 of the maize crop and were 162 ± 6 t C ha^{-1} (mean and standard error). In contrast, the pasture contained occurrences of all four soils but was dominated by the Te Puninga silt loam (Mottled Orthic Allophanic (Hewitt, 1998)). The Te Puninga silt loam was a moderately gleyed imperfectly drained soil (McLeod, 1992) with a measured bulk density and carbon content of the topsoil of 905 kg m^{-3} and 9.3 % respectively. The permanent wilting point of the Te Puninga silt loam was 0.25 m^3 m^{-3} (Rutledge et al., 2017a). A total carbon stock (to 0.6 m) of 135 ± 15 t C ha^{-1} was measured for the pasture paddock. The maize and pasture sites were approximately 1.5 km apart and thus experienced the same climate. Mean annual temperature and rainfall (1981–2010) for the region were 13.3 °C and 1249 mm (NIWA, 2018).

2.1.2. General farm management

Troughton Farm is a rotationally grazed dairy farm totalling 199 ha in size and milking an average of 525 Jersey dairy cows. Approximately 13 ha of the farm (6.5 %) was cropped with maize (*Zea mays* L.) for silage annually. Paddocks in which the maize crop was grown underwent a general pattern of preparation and seeding in the spring, harvest in the early autumn before being returned to either permanent pasture or winter crop of annual pasture. Winter crops were planted between successive maize crops in the same paddock, and permanent pastures were sown when the maize cropping of individual paddocks had concluded. Maize crops were grown for a maximum of four years (but typically fewer) before being returned to permanent pasture. The remainder of the farm was pasture, which was rotationally grazed by the dairy cows year-round. Grazing events were typically 12 − 24 h in duration but extended to several days during the non-lactation winter period. Nutrient application to the pasture was via fertiliser, sludge, and compost application, and resulted in generally low annual application rates. For the three calendar years covering our study period (2016–2018), these averaged 37 kg N ha^{-1} y^{-1}, 3 kg P ha^{-1} y^{-1} and 20 kg K ha^{-1} y^{-1}. Management events for P31 during the study event are listed in Table S 1.

2.1.3. Maize crop management

Crop management for both measurement years was similar. Paddock preparation began in September with the removal of remaining biomass through harvesting (year 1 only) and grazing before killing the remaining sward with herbicide. Amendments including liquid effluent, solid effluent, compost, and synthetic fertiliser were added before tillage using a disk ripper and rototiller. The maize seeds (Pioneer® brand hybrid P0891) were planted with a second application of synthetic fertiliser. The length of preparation time differed for both years (year 1–21 days; year 2–30 days), but the planting date was very similar (14 October 2016 and 15 October 2017). Additional herbicide was applied to control weeds approximately five weeks after planting, and a third fertiliser application was made six to seven weeks after planting. Harvesting of the maize crop occurred in March (early autumn) before the establishment of either a winter crop (year 1) or permanent pasture (year 2). Following harvest in year 2, the maize paddocks were grazed to remove any spilled maize silage and to remove any grass in areas of the paddocks where maize was not sown. The winter crop/permanent pasture was established by direct drilling followed by rolling of the paddock. Finally, fertiliser was applied to the winter crop/permanent pasture in late-April/early-May before the first grazing of the pastures during May. A full description of the dates and application rates (where applicable) of these management events is available in Table S 2, while a summary of the nutrient applications during the cropping process can be found in Table S 3.

For this study, we have chosen to focus on the period between the beginning of maize crop establishment (defined as initial herbicide application) through to return of the paddocks to containing a pasture sward (defined as pasture seedling emergence). Confining our study to this period allowed for a direct comparison between the maize silage crop and the alternative use of grazed pasture.

2.2. Net ecosystem carbon balance

Calculation of the net ecosystem carbon balance (NECB) for the two study sites followed the same general process of accounting for all imports and exports of C within a defined system boundary. For the pasture site, the system boundary was defined by the fence line surrounding P31, while for the maize site this was defined as the boundary surrounding both P58 and P60. Calculation of NECB followed the same generic equation:

$$NECB = NEP + C_{Import} - C_{Export} \quad (1)$$

Expanded to account for all import and export terms this equation takes the form:

$$NECB = NEP + F_{Manure} + F_{Effluent} + F_{Fertiliser} + F_{Supp} + F_{Excreta\ returned} - F_{Grazing,P} - F_{Grazing,S} - F_{Harvest} - F_{Leaching} \quad (2)$$

Where: NEP was the net ecosystem production (equal in magnitude, but opposite in sign to net ecosystem exchange (NEE); see section 2.4); F_{Manure} was the C imported in manure (composed of solid effluent/sludge and compost), $F_{Effluent}$ was the C imported in liquid effluent, $F_{Fertiliser}$ was the C imported in synthetic fertiliser, F_{Supp} was the C in supplemental feed brought into the system boundary, $F_{Excreta\ returned}$ was the excreta (dung and urine) C deposited by grazing animals, $F_{Grazing,P}$ was the C in the biomass removed by the grazing cattle, $F_{Grazing,S}$ was the supplemental feed C consumed within the system boundary by the grazing cattle, $F_{Harvest}$ was harvested biomass C grown within the system boundary and mechanically removed, and $F_{Leaching}$ was the C leached through the bottom of the soil profile.

Eq. (2) was used for both the pasture and maize sites. However, not all terms applied to each site. For example, $F_{Effluent}$ did not apply to the pasture site during the study period, F_{Supp} and $F_{Grazing,S}$ did not apply to the maize site in either year, while $F_{Grazing,P}$ and $F_{Excreta\ returned}$ only applied in year 2. Details of the calculation method of each term for the pasture site can be found in Rutledge et al. (2017a), and while similar, the calculation methods for the maize site are described in the supplementary materials.

This study follows conventions of Chapin et al. (2006), where the carbon fluxes (NEP, NECB, and its components) use the sign convention of positive values representing a gain to the ecosystem, and negative values a loss from the ecosystem.

2.3. Aboveground biomass measurements

Aboveground biomass was determined using different methods for the maize and pasture. For the maize site, all aboveground biomass produced was assumed to be due to the maize crop with no plant growth during the establishment periods. Aboveground biomass from the maize site was calculated as harvested biomass (as described in the calculation of $F_{Harvest}$) plus an estimate of the residue (or stubble) remaining following harvest. To estimate the maize residue biomass > 50 plants were sampled, and then dried at 95 °C for 24 h before weighing to determine dry biomass. Total residue biomass was calculated by multiplying the average plant dry mass by the planting density. Aboveground biomass estimation for the pasture followed the procedure of Rutledge et al. (2017a) and used regular clipping of the pasture (excluded from grazing by cages).

2.4. CO_2 flux measurement and processing

Flux measurements follow the same instrumentation and processing procedures as outlined in Rutledge et al. (2017a) and Wall et al. (2020) for previous studies on the same site. The following sections briefly describe how these measurements were made, including minor differences for the maize site, with more details in the aforementioned studies.

2.4.1. Instrumentation

Eddy covariance measurements were made by combining data from a 3D sonic anemometer (CSAT3; Campbell Scientific Inc., Logan UT USA), and an infrared gas analyser (LI-COR Inc. Lincoln, NE, USA). For the maize site, the infrared gas analyser was an open path LI-7500A, while for the pasture site an enclosed path LI-7200 was used. Measurements were made at a height of 1.55 m for the pasture, and began at 1.61 m for the maize but were adjusted several times during the growing period to ensure sufficient distance between the canopy and instrumentation. Data were collected at 20 Hz using a CR3000 datalogger (CSI) and stored on a compact flash card.

Configuration and instruments used to collect ancillary measurements were the same at both sites. These measurements included volumetric moisture content (VMC; CS616, CSI) and soil temperature (107-type probe, CSI) measured at 50, 100 and 200 mm depths, soil temperature using a four-junction averaging thermocouple (TCAV, CSI) with probes at 20 and 60 mm depths, soil heat flux (HFP01, Hukseflux Thermal Sensors, Delft, Netherlands) at 80 mm depth, air temperature and relative humidity (HMP155, Vaisala, Helsinki, Finland), and rainfall (TB3, Hydrological Services). Additionally, a four-component net-radiometer (NR01, Hukseflux Thermal Sensors) was used to determine short- and longwave radiation. Like the EC instrumentation, the measurement height for the net radiometer at the maize site was adjusted according to the canopy height. All ancillary data were measured at 1 Hz, averaged or totalised at 30-minute intervals and recorded on either a CR1000 or CR3000 datalogger (CSI).

2.4.2. Flux processing

Where possible, processing of the EC data to calculate the net ecosystem exchange (NEE) of both the maize and pasture sites were standardised. Differences occurred due to the different gas analysers used at each site. Flux data were calculated from the 20 Hz raw data using EddyPro version 6.2.2 (EddyPro ® Version 6.2 [Computer software], 2016). Within EddyPro, raw data were screened for spikes, amplitude resolution, dropouts, absolute limits, and skewness and kurtosis following Vickers and Mahrt (1997). Axis rotation for tilt correction of the wind data followed the double rotation method, while data were detrended using block averaging. Any time lag between the anemometer and infrared gas analyser variables was determined using covariance maximisation (maize site) or using the EddyPro automatic time lag detection method to account for air sample travel time (pasture site; LI-COR, 2016). Finally, 30-minute processed fluxes were calculated using molar mass densities (maize site) or mixing ratios (pasture site) of CO_2 and H_2O. The WPL correction (Webb et al., 1980) was applied to the fluxes calculated for the maize site. Additional corrections to both datasets included analytic correction for high-pass filtering effects (Moncrieff et al., 2004), in-situ/analytic correction of low-pass filtering effects (Fratini et al., 2012), and instrument separation (Horst and Lenschow, 2009).

2.4.3. Filtering and gap filling

The processed flux data were filtered according to several criteria before the gap-filling step. Half-hourly data were rejected during periods of instrument malfunction, when instrument warnings exceeded a set threshold (0.5 % of individual measurements in a half-hour), and during site visits. Unrealistically low or high fluxes (< -80 μmol m^{-2} s^{-1} and > 35 μmol m^{-2} s^{-1} for maize, and < -36 and > 18 μmol m^{-2} s^{-1} for pasture) and those half-hours where Mauder and Foken (2004) quality control flags resulted in a value of 2 (poor quality) were also rejected. Only fluxes collected under conditions of sufficient turbulence, defined as the standard deviation of the vertical wind speed (σ_w; Acevedo et al., 2009) exceeding 0.10 m s^{-1} (determined by adapting the methods of Reichstein et al. (2005)), were retained. Additional filtering criteria excluded flux data collected when the mean wind direction of the half-hour originated from behind the supporting tower, and the standard deviation of the wind direction was either too large (> 40) or too small (< 0) (Hunt et al., 2016). Flux footprint analysis (Kormann and Meixner, 2001) was used to ensure only fluxes representative of the measurement area were used. During year 1, occasional measurements of turbulent energy fluxes from the maize site significantly exceeded available energy, and thus half-hours where the turbulent energy exceeded available energy by more than 200 W m^{-2} were also rejected. Finally, we chose to exclude all half-hours where cattle were grazing an area that contributed more than 5 % to the flux footprint (primarily of

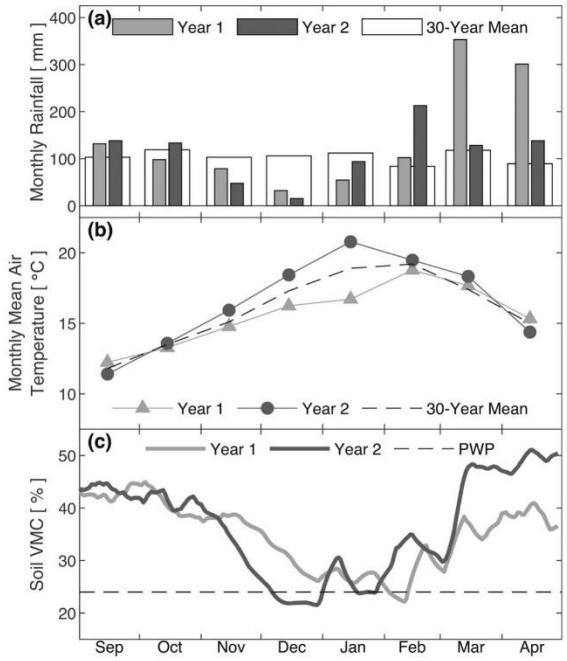

Fig. 2. Climate variables for two maize cropping seasons as measured at the maize site: (a) measured monthly rainfall including 30-year monthly mean rainfall; (b) monthly mean air temperature; and (c) 10-day running mean volumetric soil moisture content at 100 mm depth, and indicative permanent wilting point (PWP) as measured at a nearby site on the same soil type.

importance to the pasture site). After the filtering step for the two measurement periods, remaining high-quality data coverage was 40.5 % and 48.0 % at the maize site, and 32.2 % and 34.3 % at the pasture site.

An artificial neural network (ANN) approach (Papale and Valentini, 2003) was used to fill gaps. Both the maize and pasture sites were filled in the same manner using the same driving variables. A detailed description of the gap-filling method is available in Wall et al. (2020) and is summarised in brief here. Day- and night-time data were gap-filled separately using different driving variables. For the daytime NEE model, input variables to the ANN included photosynthetic photon flux density (PPFD), vapour pressure deficit, air temperature, soil temperature (from the four-junction averaging thermocouple), volumetric moisture content (100 mm depth) and ΔNEE. ΔNEE is the non-normalised phytomass index (Aurela et al., 2001), which is an ideal descriptor for estimating above-ground photosynthetic material in grazed ecosystems (Campbell et al., 2015). We found ΔNEE worked equally well as an input driver for the duration of the maize cropping cycle (including both the cultivation and growth periods). The nighttime ANN had input drivers of air temperature, soil temperature (from both the four-junction averaging thermocouple and 100 mm depth), volumetric moisture content (100 mm) and ΔNEE. For half-hours where measured values were available, the ANN modelled values compared well (r^2 of > 0.95 for both maize and pasture sites).

Partitioning of ecosystem respiration (ER) and gross primary productivity (GPP) from NEE was undertaken by firstly assuming that GPP was zero during the night and thus all nighttime CO_2 flux was ER. The nighttime ANN model was then extended to run during the daytime utilising the same driving variables to provide a complete time series of ER. Finally, GPP was calculated as ER – NEE. GPP and ER are reported as positive values. Both NEP and ER exclude grazer respiration with fluxes measured during grazing excluded from analysis (see above).

2.5. Uncertainty analysis

Uncertainty analysis followed the same procedure as originally described by Rutledge et al. (2017a) and Wall et al. (2020). Our uncertainties are presented as 95 % probability intervals using an approach inspired by Bayesian elicitation (O'Hagan et al., 2006). Firstly, probability distributions (described below) were determined for uncertainties around all NECB components. A Monte Carlo simulation was used to produce 1000 estimates of each component. All calculations were then repeated for each simulation run to produce 1000 (for example) NECBs. The uncertainty was then determined as the 2.5^{th} and 97.5^{th} percentile values from the simulations, often resulting in asymmetrical uncertainties. Differences between datasets were defined as non-overlapping probability intervals.

Derivation of the probability distributions required estimates of the mean and standard deviation of each component (e.g. for manure, this included mass applied, dry matter content, and carbon content). We assumed our measurements (or estimate in the case the harvested maize biomass) were equivalent to the mean. For NEE, the standard deviation was calculated by summing in quadrature the uncertainty associated with the random uncertainty (measurement and gap-filling; following Hollinger and Richardson, 2005; Dragoni et al., 2007), choice of turbulence (σ_w) threshold, and the ANN gap-filling process (Wall et al., 2020). For all other components, the standard deviation was calculated from the analysis of multiple samples, or if unavailable, from an estimate of the plausible range of values. Where an estimate was made, the plausible range of values was conservatively assumed to account for 95 % of the variance, and the standard deviation was calculated as one-quarter of this range. The result was large uncertainties applied to data where confidence was lower (e.g. harvested biomass), and much smaller uncertainties where values were much more constrained (e.g. fertiliser applied mass, biomass carbon content). Additional details of the derivation of the probability intervals for the non-NEE components can be found in (Rutledge et al., 2017a).

3. Results

3.1. Climate

During the period of interest (the maize cropping season; September to March), year 1 (2016/17) was overall cooler (mean air temperature of 15.6 °C) and wetter (total rainfall of 851 mm) than the 30-year normal (16.2 °C and 745 mm respectively; NIWA, 2018), while year 2 (2017/18) was warmer (16.8 °C) with near-normal rainfall (769 mm). For both years, rainfall followed a similar pattern of near-normal in September and October, considerably drier from November to January before deviating in the February to April period (Fig. 2a). The rainfall pattern was reflected by the soil volumetric moisture content (VMC) (Fig. 2c). Both years had a similar soil VMC through September and October, however, the soil dried quicker in year 2 resulting in VMC being below the permanent wilting point for most of December 2017, and at PWP for approximately half of January 2018. In contrast, VMC only decreased below PWP for a short period in year 1 (February 2017). Air temperature was near normal in all months of both years except for November and December, which were cooler than normal in year 1 and warmer than normal in year 2 (Fig. 2b).

3.2. Flux totals and biomass production

The CO_2 dynamics of the maize and pasture sites differed through the maize cropping season (Fig. 3). GPP of the maize crop was controlled by the presence and growth stage of the maize plants, while ER was much more consistent. This combination resulted in the NEP of the maize site being negative (net loss of CO_2) for extensive periods and indeed daily GPP did not exceed daily ER until several weeks after planting. Contrastingly, both GPP and ER of the pasture site were much

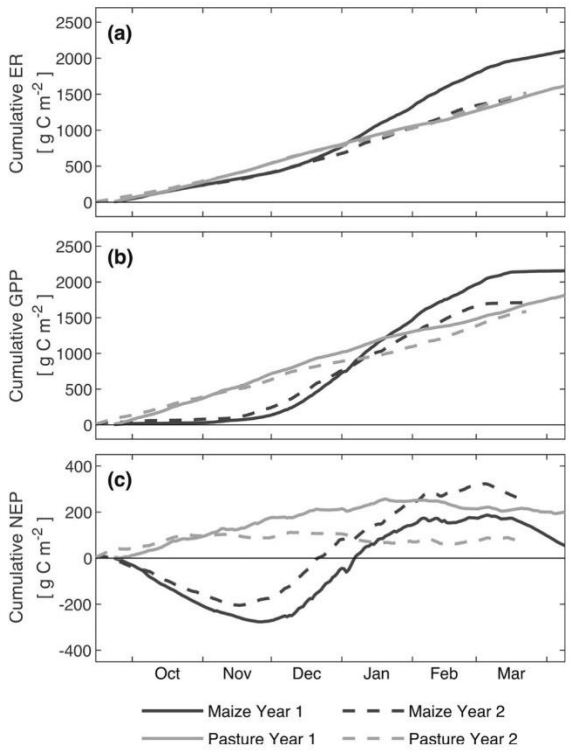

Fig. 3. Cumulative respiration (ER; a), gross primary productivity (GPP; b) and net ecosystem production (NEP; c) for maize and pasture during year 1 (2016/17) and year 2 (2017/18) cropping periods.

more consistent, and variation in NEP more subtle. However, the summed fluxes (GPP, ER, and NEP) to the end of the cropping seasons were not as dissimilar. Summed GPP and ER from year 1 for the maize site were 2151 and 2096 g C m^{-2} respectively, and were much larger than the pasture site (GPP was 1804 g C m^{-2} and ER was 1607 g C m^{-2}). In year 2 the maize and pasture GPP and ER were more similar, with GPP of the maize (1702 g C m^{-2}) higher than the pasture (1578 g C m^{-2}), but ER of the maize (1462 g C m^{-2}) slightly lower than the pasture (1512 g C m^{-2}). Comparison between the two years is slightly more difficult due to the longer cropping season of year 1 (198 days) compared to year 2 (189 days). Nonetheless, both GPP and ER of the maize was considerably larger in year 1 than year 2, while the larger totals in year 1 of the pasture relative to year 2 are mainly due to the longer cropping season. Had the first year concluded after 189 days (length of year 2) ER of the pasture would have been almost identical, while GPP of year 1 would have been slightly higher (Fig. 3).

Unsurprisingly, aboveground biomass, like GPP, for both the maize and pasture crops were larger in year 1. Estimated aboveground biomass for the maize was 26.7 t DM ha^{-1} in year 1, and 21.0 t DM ha^{-1} in year 2, and was significantly lower for the pasture at 7.9 t DM ha^{-1} and 7.4 t DM ha^{-1} for the two measurement periods. The decrease in maize production in year 2 (79 % of year 1) was matched by an equivalent percentage decrease in GPP. Relatively, the pasture performed slightly better than the maize in year 2 with aboveground biomass and GPP being 94 % and 87 % compared to year 1 values. The poorer biomass production in year 2 was likely explained by most of the December and January months experiencing soil moisture limitations (Fig. 2c). This moisture limitation occurred during the key growing period of the maize crop (Fig. 3b), and thus may have had a proportionally larger impact on the maize aboveground biomass compared to the pasture.

After the cropping season in year 1, the summed NEP of the maize was a small CO_2 sink of 55 g C m^{-2}, which contrasted with the much larger sink of the pasture (197 g C m^{-2}). The opposite pattern occurred in year 2 with the maize being a stronger sink (240 g C m^{-2}) than the pasture (66 g C m^{-2}). Averaging both cropping seasons yielded similar NEP for maize and pasture at 147 and 132 g C m^{-2} respectively. The variation between the years was likely due to a combination of different climates and crop season lengths, having differing effects on the maize and pasture. The total accumulated NEP for the maize cropping season was dependent on the definition of the cropping season. In this study, we defined the cropping season as the period from the last day in pasture (date of herbicide application) to the emergence of the pasture seedlings. For the actual period the maize crop was growing (i.e. from planting to harvest), NEP of the maize for the two seasons was 287 and 415 g C m^{-2} (Table 1), with considerably smaller net CO_2 uptake than when the preparation periods were included. Indeed, the preparation period before planting of the maize resulted in losses of –114 and –96 g C m^{-2} (years 1 and 2 respectively), while the period between harvest and re-planting with pasture lost a further –69 and –8 g C m^{-2}, primarily through lack of photosynthetic inputs (Table 1).

3.3. Carbon budgets

The NECB was calculated for both the maize and pasture sites for

Table 1
Accumulated net ecosystem production (NEP) between different events (see Table S 2 for dates of events). Each cell in the matrix displays the accumulative NEP between the event specified by the row label and the column header. The upper of the two values is for year 1 (2016/17), and the lower value is for year 2 (2017/18). Positive values represent a gain to the ecosystem (net uptake), while negative values indicate the ecosystem being a net source of CO_2. All values have units of g C m^{-2}.

	Planting date (Maize)	Seedling emergence (Maize)	Daily GPP first exceeds daily ER[1]	Harvest date	Planting date (Pasture)	Seedling emergence (Pasture)
Spray date	−114	−177	−276	173	104	55
	−96	−141	−207	318	310	240
Planting date (Maize)		−63	−162	287	218	169
		−45	−110	415	407	336
Seedling emergence (Maize)			−99	350	281	232
			−65	460	452	381
Daily GPP first exceeds daily ER[1]				449	380	331
				525	517	446
Harvest date					−69	−118
					−8	−79
Planting date (Pasture)						−49
						−71

[1] Daily GPP exceeded daily ER in year 1 on 27 November 2016 and in year 2 on 18 November 2017.

Table 2
NECB and components for maize and pasture according to Eqs. (1) and (2). The measurement period of each year extended from the time of initial preparation for the maize crop to seedling emergence of the pasture (23-Sep-2016 to 9-Apr-2017 for year 1, and 15-Sep-2017 to 31-Mar-2018 for year 2). Units are in g C m^{-2}, and positive values indicate gains of C to the ecosystem. Uncertainties in square brackets are asymmetrical 95 % probability intervals. Columns may not total due to rounding.

	Maize		Pasture	
	Year 1	Year 2	Year 1	Year 2
NEP	55 [–50/+48]	240 [–32/+31]	197 [–21/+22]	66 [–24/+25]
F_{Manure}	164 [–26/+27]	153 [–38/+41]	0	0
$F_{Effluent}$	0	9 [± 3]	0	0
$F_{Fertiliser}$	13 [± 1]	7 [± 1]	6 [± 1]	0
F_{Supp}	0	0	8 [± 2]	4 [± 3]
$F_{Excreta\ returned}$	0	3 [± 0]	119 [± 9]	89 [–7/+6]
$F_{Grazing,P}$	0	–5 [± 2]	–311 [–31/+29]	–222 [–19/+16]
$F_{Grazing,S}$	0	0	–6 [± 1]	–3 [± 2]
$F_{Harvest}$	–1081 [–192/+165]	–820 [–131/+128]	0	–47 [± 16]
$F_{Leaching}$	–2 [± 1]	–1 [± 1]	–1 [± 1]	–1 [± 1]
C_{Import}	*177 [–26/+27]*	*171 [–39/+41]*	*133 [± 9]*	*93 [± 7]*
C_{Export}	*–1083 [–192/+165]*	*–826 [–131/+127]*	*–319 [–31/+29]*	*–273 [–25/+22]*
NECB	**–850 [–188/+169]**	**–415 [–146/+141]**	**11 [–39/+37]**	**–114 [–37/+34]**

the period between herbicide application before maize establishment and seedling emergence of the subsequent pasture sward (Table 2). In both years, there were large C losses with the NECB for maize being –850 g C m^{-2} in year 1 and –415 g C m^{-2} in year 2. The pasture site was near neutral (11 g C m^{-2}) in year 1, and source of C in year 2 (–114 g C m^{-2}). All four site-years were significantly different from each other (Table 2). Harvested material represented the single largest component in the carbon balance of the maize representing the removal of 1081 g C m^{-2} (year 1), and 826 g C m^{-2} (year 2). This harvest biomass equated to > 90 % of all aboveground biomass grown, with only ~2 t DM ha^{-1} (~85 g C m^{-2}) remaining as stubble (residue). In contrast, while still the largest contributing component to the NECB, biomass removed from the pasture via grazing ($F_{Grazing,P}$) and harvest ($F_{Harvest}$; year 2 only) totalled only 311 g C m^{-2} (year 1) and 269 g C m^{-2} (year 2). The proportion of biomass removed relative to that grown was greater for the maize than the pasture. For the pasture system we used a utilisation value of 0.85 (DairyNZ, 2017), thus assuming that the remaining 15 % of grown biomass remained in the paddock as residue, and therefore proportionally more aboveground biomass of the maize site was exported. Biomass removal for both the maize and pasture constituted almost exclusively the total C exports for the ecosystems with small additional losses through leached C ($F_{Leaching}$). We note that $F_{Leaching}$ from the maize site may be underestimated due to maize roots potentially extending deeper than the 0.6 m used in the calculation.

Imports of C to the ecosystems (excluding NEP) were considerably smaller than exports for both the maize and pasture. Carbon imported to the pasture ecosystem was primarily excreta (dung and urine) deposited by the grazing animals. Other imports included synthetic fertiliser and supplemental feed, but the totals of these during the measurement period were small (14 g C m^{-2} in year 1, and 4 g C m^{-2} in year 2). In the maize system, manure (sludge/solid effluent and compost) was the major import of C, which when added to (liquid) effluent was applied in similar quantities in both years (164 and 162 g C m^{-2}). Small amounts of C in fertiliser were also added, while in year 2, a grazing event immediately following harvest resulted in a small amount of deposited excreta (Table 2).

For both the maize and pasture, NEP was a major net input of carbon to the ecosystems. NEP was the largest input of C to the maize system in year 2, and the pasture system in year 1. On a day-to-day basis, NECB of the maize ecosystem was primarily controlled by NEP, with only the occasional irregular C import and export events (e.g. manure application, and harvest) causing an offset to what was essentially cumulative NEP (Fig. 4). The large import (manure application) and export (harvest) events had a major effect on NECB causing rapid switches from the maize ecosystem being a large sink of C to a large source of C almost instantaneously. For example, immediately before harvest in year 2, the maize site was a sink of about 500 g C m^{-2}, then within the time taken to harvest the biomass (~1 h), the ecosystem had become a source of about –300 g C m^{-2} (Fig. 4). In contrast, day-to-day variation in the pasture NECB was much more constrained (relative to the maize), with more frequent grazing events resulting in smaller regular imports and exports of C. The differing management and growth behaviour of the pasture and maize systems resulted in times when each had a greater carbon budget than the other, and therefore differences between the two ecosystems were dependent on the period of interest.

Finally, total soil C stock (to 60 cm) for the maize site, as measured between the first and second measurement years, was 162 ± 6 t C ha^{-1}, and thus the carbon balance for the first measurement year was equivalent to a loss of ~5 % of the total C stock. For year 2, the carbon budget of –415 g C m^{-2} (–4.15 t C ha^{-1}) was equivalent to another decrease in soil C stocks of ~2.6 %. Across the two years, the combined carbon balance from maize silage production equates to about 7.5 % of the C stock. In contrast, the loss of –114 g C m^{-2} (–1.14 t C ha^{-1}) from the pasture site in year 2 was equivalent to a change of ~1 % in total soil C stocks of that site.

4. Discussion

In this study, we compared the carbon balance of maize silage cropping for two years with a rotationally grazed pasture over the same period. Both maize silage crops resulted in negative carbon balances of –850 g C m^{-2} (year 1) and –415 g C m^{-2} (year 2) and contrasted with the pasture carbon balance of 11 g C m^{-2} in year 1, and –114 g C m^{-2} in year 2. All site years were significantly different from each other. We chose to restrict our study to just the period of the maize crop for two reasons. Firstly, to provide a direct comparison with the alternative use of permanent pasture during the cropping period, and secondly because annual balances are strongly impacted by the choice management during the winter period, which for our site differed in each of the study years. The following discussion focuses on the carbon budget of the maize silage crop, and how it contrasted with the pasture, which was in line with longer-term records for the site (Wall et al., 2020).

4.1. Carbon balance of maize silage cropping

Conversion of pasture to maize silage cropping resulted in negative carbon balances (averaging –632.5 g C m^{-2}) from the period of initial crop establishment through to seedling emergence of re-established pasture following maize harvest. In the few similar studies including

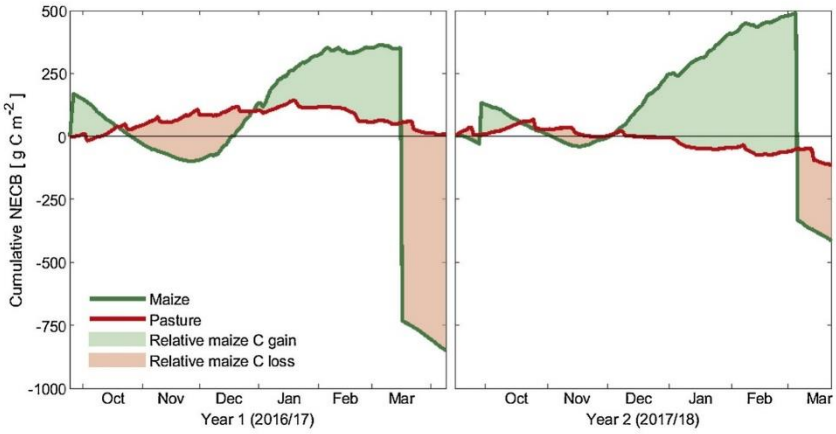

Fig. 4. Cumulative NECB for both the maize and pasture sites during each cropping season. The filled colours indicate the difference in ecosystem carbon between maize and pasture, with green indicating the maize site having a greater carbon balance than the pasture, and the red the opposite (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article).

full biomass removal (i.e. grains and stover), similar large losses were observed. Poyda et al. (2019) reported carbon balances from multiple cropping fields over several years from Germany, which included some years where maize silage was grown. Of all the crops that their measurements included, maize silage was the largest source of C with carbon balances ranging from around –200 to –800 g C m^{-2} y^{-1}. Likewise, Béziat et al. (2009) found a maize silage crop in France to be a C source of –372 g C m^{-2} y^{-1} and was the largest C source of eight different crops considered. While not grown for maize silage, Eichelmann et al. (2016) reported a carbon balance from a Canadian biofuel corn crop of –699 g C m^{-2} y^{-1} with all biomass removed. Worthwhile to note is that these studies were all from long-term cropping systems and reported on an annual basis (i.e. not entirely due to the maize crop alone), and therefore are not directly comparable to our study.

In a New Zealand study reporting long term C stock change under maize silage cropping using a repeated soil sampling approach, Shepherd et al. (2001) reported C losses ranging from –240 to –330 g C m^{-2} y^{-1}. In contrast, our two-year carbon balances for the maize crop were an average C source of –632.5 g C m^{-2}. Assuming the losses are representative of a change in soil C, our higher rates of loss relative to those reported by Shepherd et al. (2001) may be explained in several ways. Firstly, Shepherd et al. (2001) found that the rate of loss decreased with increased time under maize, which for our site were years three and four (P58), and one and two (P60). At one site, Shepherd et al. (2001) reported the rate of loss decreased from –240 g C m^{-2} y^{-1} for the first 11 years to 80 g C m^{-2} y^{-1} for the next 10 years, while at a second site the rates of loss went from –330 g C m^{-2} y^{-1} after 4 years to –90 g C m^{-2} y^{-1} after 11 years. Secondly, we reported the NECB for the maize cropping period itself, while the sites studied by Shepherd et al. (2001) were on an annual basis, including the winter (non-cropping) period where the land reverted to grazed pasture. Rutledge et al. (2015) showed that vigorous growth and CO_2 uptake of new pastures in New Zealand systems could overcome a period of significant C loss during establishment, therefore, indicating new pastures to be C sinks. Consequently, the inclusion of the winter pasture period (and thus a potential C sink) of our site may partially mitigate for the C loss associated with the maize crop, and thus move the annual C losses closer to those of Shepherd et al. (2001). Finally, if we consider the pasture site to act as a control (i.e. had the maize crop not been grown), the C source strength due to the maize cropping may indeed be smaller as the summed carbon balance of the two measurement periods for the pasture site was –103 g C m^{-2}.

4.2. Importance of biomass to the carbon balance

Biomass removal strongly shapes the carbon balance of both the maize and pasture systems. Aboveground biomass production from the maize crop was around three times greater than that of the pasture during the same period despite the GPP of the maize being only 10–20% higher. A much lower root:shoot ratio of the maize (~0.18; Bolinder et al., 2007) than the pasture (0.75–1.5; Kirschbaum et al., 2017) results in the maize being much more efficient at producing aboveground biomass, which was then available for harvest. Biomass (and thus C) removal of maize crops can vary from approximately half in grain-only systems (Eichelmann et al., 2016) to greater than 90 % with full removal (e.g. this study), while pasture systems are generally much more consistent. When biomass is grown for direct animal consumption (whether pasture or maize silage), the ultimate goal is to maximise the usage of that feed. For the maize silage in our study, this resulted in 92 % of all aboveground biomass being exported for later usage as supplement feed. During the grazing of pasture, a certain proportion of the pasture is not consumed due to trampling by the grazers or covered by animal excreta deposition. Wastage in well-managed New Zealand pastures is estimated at 15 % (DairyNZ, 2017) and thus only about 85 % of aboveground biomass is eaten by the grazers. Consequently, proportionally less biomass is immediately removed from the pasture system than the maize system. However, in the pasture system, the grazing animals also cycle about a third of ingested biomass as excreta, much of which is deposited on the grazed paddock. Due to the short grazing duration (average of 12 h) coupled with the digestion time of the animals (~24 h for pasture; Moran, 2005) much of the excreta deposited on the pasture measurement site, resulted from pasture and supplemental feed eaten outside the measurement paddock. Nonetheless, we can assume that about one-third of all grazed pasture C is returned to the paddock as excreta, which when coupled with 15 % wastage, yields a net of only around 60 % of the grown pasture C being exported from the paddock, predominantly as respiration and product (milk), compared to > 90 % for the maize. This lack of carbon cycling by grazing animals coupled with the more efficient production of aboveground biomass in a maize silage system, therefore, contributes to the much larger observed carbon loss.

4.3. Mitigation potential of maize silage cropping

Despite observing large losses of C during the maize cropping period, there may be potential for the production of maize silage to increase soil C stocks when considering a broader carbon cycle at the farm rather than paddock scale. Conversion of cropland to pasture is generally associated with an increase in soil C stocks (Conant et al.,

2017), and indeed New Zealand maize cropping systems have shown increased soil C stocks following a return to pasture (Shepherd et al., 2001). In a dairying system such as that of our study farm where maize cropping is rotated around the farm with each maize paddock being returned to permanent pasture within a few years, it is plausible that any lost soil C could be recovered before the paddock is returned to maize. For example, our study farm utilised 6.5 % of the land area annually for maize silage, and thus a return period for maize cropping would be around 15 years if each paddock were only cropped for one year at a time. Assuming a recovery rate under pasture of about 90 g C m^{-2} y^{-1} (Shepherd et al., 2001; Conant et al., 2017), and an average annual loss of ~630 g C m^{-2} (as per our study), the lost soil C from a single cropping event could potentially be recovered within about seven years. This time frame is well within the 15-year return time of the maize crop, and close to the 7-year maize silage/permanent pasture rotation suggested by Densley et al. (2001) to improve animal feed availability and thus productivity. Therefore, it is plausible that while a short-term C source, the production of maize silage may be C neutral longer term although too frequent a return period would result in a gradual long-term decline in C stocks.

Additionally, assuming longer-term C neutrality during the production of maize silage within a rotationally cropped dairy system, use of the produced supplement feed within the same system allows for sequestration of C. In consuming the harvested biomass, the grazing animals then excrete approximately one-third of the ingested C, much of which will be deposited on the grazed pastures. Manure (excreta) sequestration rates are estimated at around 12 % (Maillard and Angers, 2014), and therefore about 4% of all harvested biomass may eventually be sequestered. For our study farm, annual production of maize silage yields an additional ~20 g C m^{-2} y^{-1} of excreta deposition across the entire farm and thus allowing for a small farm-wide gain of ~2.5 g C m^{-2} y^{-1}. While the gain is small, it is cumulative provided on-going use of maize silage thus leading to larger long-term gains. Indeed, the single process of supplement feed (i.e. maize silage) usage has been suggested to lead to gains in soil C by modelling (Kirschbaum et al., 2017) and experimental approaches (Wall et al., 2019). Further benefits of shifting maize silage production to within a rotationally cropped dairy system could include a reduction of continuously cropped land (allowing the potential for increasing soil C stocks of that land), and production of higher-yielding crops due to higher initial soil C contents (Lal, 2011).

While we argue the potential of C neutrality for periodic maize silage cropping, this is still uncertain and therefore any reduction in the loss would be beneficial. The magnitude of these losses may be able to be reduced by modifying management practices including establishment time, tillage intensity, and C amendments. A reduction in establishment time, and thus limiting the time with no photosynthetic inputs, for either the maize crop or the return to pasture following harvest would allow for reduced C losses, similar to recommendations made by Rutledge et al. (2017b) for pasture renewal. Indeed, the pasture establishment following the harvest of the maize in year 2 provides an example as it was undertaken much quicker than year 1 (3 days vs 14 days), resulting in smaller CO_2-C losses during this period (Table 1). Secondly, a reduction in tillage intensity from full cultivation may also be beneficial (Lal, 2011), especially as a corresponding reduction in establishment time may result. Finally, the addition of more C inputs such as organic manure would further decrease the net C source strength. However, the addition of such amendments may be contradictory to a reduction in establishment time and tillage intensity, as they are often ploughed into the soil during the cultivation process (e.g. this study).

Finally, the use of maize silage as supplemental feed may have conflicting impacts on the other greenhouse gases nitrous oxide (N_2O) and methane (CH_4). The measured nitrogen content of the maize silage (1.2 %) was considerably lower than that of pasture (3.6 %; Wecking et al., 2020), and given N_2O emissions in pastoral systems are tied to the N content of the consumed animal feed (Dijkstra et al., 2011), feeding of the maize silage may result in a reduction in N_2O emissions. While New Zealand reports GHG emissions of enteric CH_4 on a basis of total animal feed intake irrespective of the feed type (MfE, 2019), there is some evidence to suggest enteric CH_4 emissions from maize silage may be lower than for pasture (van Gastelen et al., 2015; Dall-Orsoletta et al., 2019). Consequently, if maize silage were to replace pasture on a DM matter basis (i.e. 1 kg DM of maize replaced 1 kg DM of pasture), total methane emissions may be reduced. However, where maize silage cropping replaces an equivalent area of pasture, the approximately three times greater yield of the maize silage will increase total animal feed intake, and therefore total CH_4 emissions. The net result of maize silage production and usage on N_2O and CH_4 emissions is uncertain, but if included in a full GHG budget for maize silage may have some potential to mitigate.

4.4. Constraints for interpretation

Understanding the impact of specific management practices is essential to identifying GHG mitigation options within agricultural ecosystems, and therefore careful consideration needs to be given to the period over which carbon balances are determined. Our objective was to specifically quantify the carbon balance of periodic maize silage cropping as a management option available to New Zealand dairy farmers. Because on-farm maize silage cropping within the New Zealand dairy system involves replacing pasture paddocks with maize before returning them to pasture, our period of interest extended form crop establishment through until pasture was once again re-established. While this period was required to answer our specific question, direct comparison with other studies is difficult. Most carbon balance studies of cropping systems including maize provide annual totals including the intercropping period (Ceschia et al., 2010; Qun and Huizhi, 2013; Poyda et al., 2019). In the few studies where carbon balances of the crop season alone are reported, typically only the period from establishment to harvest is included (Jans et al., 2010; Wang et al., 2015). By including the pasture establishment period, we calculated a larger C source than at the point of harvest (Table 1), however, if extended to produce an annual total further modification would be expected due to the combined effects of the new pasture/winter crop (Rutledge et al., 2015) and grazing management (Wall et al., 2020). This example highlights the importance of the precise determination of the measurement period to isolate the effect of individual management practices.

A second factor influencing the interpretation of our study was the large harvested maize biomass uncertainty. Neither method used to quantify the harvested biomass was a direct measurement, which would have provided much-improved certainty of C exported from the paddock. Nonetheless, the key message is that periodic maize silage cropping within a grazed pastoral system resulted in large C losses in the order of 400 − 850 g C m^{-2} per crop. Despite the large uncertainties (almost ± 190 g C m^{-2} for year 1 of the maize site), our study indicates the magnitude of C lost, which therefore needs to be recovered for maize silage supplemental feed within New Zealand dairy farms to be considered C neutral.

5. Conclusions

Production of maize silage as a supplemental feed from a periodically cropped grassland of a New Zealand dairy farming system was a larger source of carbon relative to the pasture alternative. During the cropping period a carbon balance of –850 g C m^{-2} in year 1, and –415 g C m^{-2} in year 2 was calculated for the maize silage, and 11 g C m^{-2} (year 1) and –114 g C m^{-2} (year 2) for the pasture. This difference between the maize and pasture systems was primarily due to much higher total biomass production of the maize site (three times that of the pasture), coupled with a lower proportion of aboveground biomass removed in the pasture system. Consumption of the maize silage by

grazing animals within the same farm system led to additional excreta deposition across the grazed areas (including the pasture site) and was estimated at ~20 g C m^{-2} y^{-1} for the study farm. Periodic maize silage cropping resulted in increased annual feed but at a cost in the order of ~630 g C m^{-2} per crop, which requires recovery before any future cropping to avoid a long-term decline in C stocks. Further work would be beneficial to (1) investigate the recovery of ecosystem C following re-establishment of permanent pasture following maize silage cropping, (2) consider the options aimed at minimising the magnitude of the loss in these systems, and (3) perform a full greenhouse gas assessment.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supplementary data

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Chapter 5:

Temperate grazed grassland carbon balances for two adjacent paddocks determined separately from one eddy covariance system

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Please refer to Appendix D for the contribution of each of the authors

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Temperate grazed grassland carbon balances for two adjacent paddocks determined separately from one eddy covariance system

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ABSTRACT

Use of eddy covariance (EC) as part of net ecosystem carbon balance (NECB) studies of grazed grasslands has the potential to provide valuable insights into the influence of management on soil carbon (C) stocks and opportunities for potential mitigation. However, in rotationally grazed systems where the EC measurements extend across multiple paddocks, often due to necessity, interpretation of results can be complicated by differing paddock-scale management regimes despite being the same land use with a single overarching farm-scale management approach. Here, we calculated separate annual CO_2 fluxes and NECBs for two adjacent paddocks under the same general land management using a single EC system for seven years. We were interested in determining whether fluxes and annual NECB of the paddocks were sufficiently similar to allow future side-by-side comparisons of management practices aimed at increasing soil C. We assigned the half-hourly flux measurements to individual paddocks (P31 and P32) using footprint analysis and then gap filled each dataset separately. Additionally, we calculated annual fluxes (and NECBs) for the integrated area of both paddocks (i.e. using all measured flux data; referred to as NewMix). The mean (± 95% confidence interval) NECB for P31 and P32 were 1 ± 76 g C m^{-2} y^{-1}, and 7 ± 80 g C m^{-2} y^{-1}, which compared well with the NECB of NewMix (–14 ± 48 g C m^{-2} y^{-1}). Large between-paddock and inter-annual variability were observed, and several consecutive years of data were required to determine with high certainty that both paddocks were C neutral. Day-to-day management decisions were identified as causing some of the variability. We concluded that the calculation of fluxes and NECB's from two adjacent paddocks using a single EC system was possible, and indeed, likely more insightful and useful than integrating across multiple paddocks. Such an approach can provide opportunities to more accurately quantify management effects, and test mitigation strategies by using treatment and control experiments on adjacent paddocks using a single EC system, although multiple measurement years are likely required. Alternatively, using two EC systems at separate sites, each quantifying both treatment and control fluxes would allow for replication, and thus greater statistical power, which has been a limitation for many EC based NECB studies.

1. Introduction

Globally, grasslands cover approximately one-quarter of the ice-free land area (Steinfeld et al., 2006), and are one of the largest terrestrial stores of soil carbon (C) (Janzen, 2004). Management practices within these grassland ecosystems have been shown to lead to both losses and gains of soil C (Conant et al., 2017). Consequently, understanding the effects of management practices on changes in soil C stocks of agricultural grasslands has been a major recent research focus, particularly within the context of using soil C to offset large-scale greenhouse gas emissions (Soussana et al., 2010; Minasny et al., 2017).

Quantification of the effect of management practises can be undertaken using different approaches. For example, changes in soil C can be measured using soil sampling techniques such as repeated sampling for temporal changes (e.g. Schipper et al., 2014) or paired site comparison for identifying differences due to management (e.g. Mudge et al., 2017). Another commonly used approach is the net ecosystem carbon balance (NECB) method of Chapin et al. (2006). Calculation of NECB has the key advantage of a higher temporal resolution allowing detection of differences on seasonal to annual timescales (Soussana et al., 2010). The NECB approach allows the ability to test the effect of management practices on grazed pastures and the subsequent C sequestration potential over shorter time scales than repeated soil sampling (e.g. Allard et al., 2007; Rutledge et al., 2014). However,

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replication in NECB studies is rare, and thus an extension of findings beyond the measurement area may be problematic (Hill et al., 2017).

The NECB method requires the measurement of all imports and exports of C to the system, including the net exchange of CO_2 between the atmosphere and ecosystem (net ecosystem exchange; NEE). Measurement techniques for NEE include the chamber method (e.g. Riederer et al., 2014) and micrometeorological techniques such as eddy covariance (EC; Aubinet et al., 2012). A key advantage of using EC is the ability to make measurements with a high temporal coverage over an integrated spatial extent in the order of hectares (Eugster and Merbold, 2015). Furthermore, within managed agricultural systems, EC allows the ability to make measurements during management events (e.g. pasture renewal, crop establishment, grazing, and harvest). EC measurements, however, require specific conditions, including sites that are homogenous and flat, with adequate fetch appropriate to the measurement height in all upwind directions (Rannik et al., 2012).

Intensively managed grasslands often have small confining areas (paddocks) which may be grazed either continuously or rotationally by animals. EC studies in such ecosystems, either by choice or by necessity, often require measurement across several paddocks (e.g. Jaksic et al., 2006; Rutledge et al., 2015; Wall et al., 2019). In these circumstances, the assumption is that the EC source area is under the same general management regime. Especially in rotationally grazed grasslands, this assumption is not necessarily true, as each paddock often has slightly different management, whether it be (for example) in the timing, duration or stocking density of grazing, or timing of amendment applications (e.g. fertiliser, manure, etc.). Consequently, these systems are not strictly homogenous and thus may violate the source area homogeneity assumption of EC (Rannik et al., 2012). Furthermore, where the measurement area includes several paddocks, measured NEE integrates across the entire area but is dominated by the paddocks (and their associated management regime) nearest to the EC system (Wall et al., 2019). Therefore, the impact that management heterogeneity has on the integrated NEE of complex grazing systems should be recognised during interpretation but is often not.

Analysis of the source footprint of EC measurements provides an avenue for developing further understanding of the effect of land management diversity within rotationally grazed grassland systems. Footprint models such as those developed by Hsieh et al. (2000), Kormann and Meixner (2001), and Kljun et al. (2015), can provide a method for isolating and assigning the measured flux to a specific source area. Such an approach has been used to determine differences in CO_2 dynamics from different land uses either side of an arbitrary boundary (e.g. Rogiers et al., 2004; Posse et al., 2014) by assigning measured fluxes to each specific land use. Similarly, footprint analysis has been used to calculate the fraction of the flux derived from a certain source area (such as a small measurement area, or different land use), and reject flux measurements failing to meet a minimum footprint fraction criterion from the source area (e.g. Schmidt et al., 2012; Ueyama and Ando, 2016; Gourlez de la Motte et al., 2018). The use of footprint modelling thus makes the evaluation of fluxes originating from heterogeneous landscapes possible.

Measurement of fluxes from a single EC system on the border of adjacent paddocks within rotationally grazed grasslands coupled to footprint analysis has the potential to allow estimation of CO_2 exchange, and by extension, C balances from each paddock. Calculation of paddock-scale fluxes allows several benefits. Primarily, the integrated signal of adjacent paddocks, each with potentially differing management, can, where flux source areas are distinct, be disentangled, and thus interpretation may be tied to specific management practices. Additionally, the influence of spatial variability on the fluxes resulting from (for example) soil type, drainage or pasture species can be limited. Furthermore, reporting of paddock-scale fluxes offers the opportunity for replication, which is rarely incorporated in EC and indeed NECB studies (Hill et al., 2017). Finally, the ability to calculate paddock-scale fluxes has the potential to allow for an experimental design that specifically tests imposed management practices against a control in adjacent paddocks. Previously, some nitrous oxide focused studies have used such methodology (Cowan et al., 2016; Fuchs et al., 2018), but NEE/NECB studies performing treatment/control investigations of grassland systems have generally used multiple EC systems (e.g. Allard et al., 2007; Ammann et al., 2007; Rutledge et al., 2017a). The use of a single EC system allows for either a reduction in the resource requirement or an increase in replication for the same resources (i.e. a control and treatment under each EC system).

To date, the impact of integrating of NEE across multiple paddocks in rotationally grazed grasslands on the interpretation of the data has not been fully explored. An improved understanding of the limitations and opportunities associated with the calculation of paddock-scale fluxes, rather than those integrated across multiple paddocks, has the potential to provide a new tool to test management related mitigation options in these ecosystems. Therefore, we aim to report NEE and NECB from two adjacent paddocks under the same management with a single EC system using footprint analysis for seven measurement years. Our objectives here are two-fold. Firstly, we compare the paddock-scale EC derived fluxes to those calculated from integrated measurements across both paddocks. This objective will test how well fluxes can be calculated for two individual paddocks using a single EC system, and furthermore identify benefits of calculating paddock-scale fluxes rather than integrated fluxes. Our second objective was to consider the temporal and spatial variability of the NEE and NECB data, and how this affects both the interpretation of reported results, and future experimental design of EC/NECB studies in rotationally grazed grasslands. If our analysis demonstrates that adjacent paddocks under similar management results in very similar NEE/NECB, this would open future opportunities for testing of proposed management practices aimed at increasing soil C at our site.

2. Methods

2.1. Site description

Data were collected from a commercial dairy farm (Troughton Farm, 37.77°S, 175.8°E, 54 m elevation), located near Matamata in the Waikato region of the North Island of New Zealand (Fig. 1) for the calendar years 2012–2018, and included four years of previously reported data (Rutledge et al., 2017a, 2017b). The experimental site consisted of two paddocks numbered 31 and 32 (P31, P32), each 3 ha in size. The combined area of P31 and P32 was referred to as NewMix by Rutledge et al. (2017a), and for consistency, we retain this name here. The pasture originally consisted of ryegrass and white clover, however, in April 2013, both P31 and P32 underwent pasture renewal. The new pasture sown was a diverse mix including grasses, legumes, and herbs. For more details of the pasture renewal process and experiment, the reader is referred to Rutledge et al. (2017b). Thirty-year (1981–2010) mean annual temperature and precipitation, measured at a nearby climate station (13 km away), was 13.3 °C and 1249 mm respectively (NIWA, 2018). All precipitation was received as rainfall, and only light frosts were experienced in winter months. Soils at the study site are a complex of silt loams formed from a parent material of rhyolitic and andesitic volcanic ashes on rhyolitic alluvium (McLeod, 1992). The soils range from well to poorly drained with the dominant soil of the study site being the Te Puninga silt loam, classified as a Mottled Orthic Allophanic soil (Hewitt, 1998). Spatial soil sampling showed P31 contained more of the better-drained soils than P32 with proportions of the dominant Te Puninga silt loam soil similar in each paddock.

The predominant wind direction at the site was from the west, and despite some seasonal differences, all wind sectors were represented year-round (Fig. 1). During the seven measurement years, wind direction data indicated P31 would have been the dominant source for flux measurements for 54.5% of all half-hours, and P32 the remaining 45.5% of half-hours

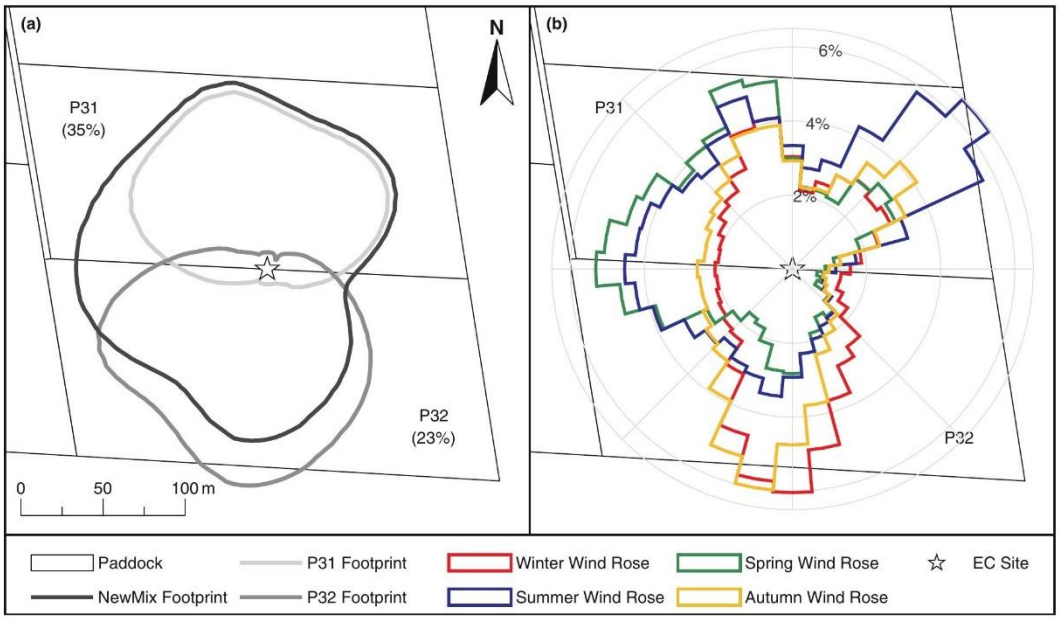

Fig. 1. Experimental field site illustrating the location of the eddy covariance (EC) system, shape, and orientation of paddocks 31 (P31) and 32 (P32). (a) 80% footprint contribution contours for NewMix, P31, and P32. Percentages in brackets beneath the paddock numbers correspond to the total contribution of the paddock to the NewMix flux over the seven measurement years. (b) wind roses for the entire seven measurement years grouped by season. The labelled percentages of the wind rose identify the percentage of time the wind originates from each direction for each season.

2.2. Farm management

Troughton Farm comprises 207 ha (199 effective ha) consisting of 67 paddocks between 2.5 and 3.5 ha in size and was rotationally grazed by Jersey cows year-round, with P31 and P32 managed in accordance with the remainder of the farm. Total cow numbers decreased each year from 701 cows in 2012 (3.52 cows ha^{-1}) to 437 cows in 2018 (2.20 cows ha^{-1}). The animal feed consisted of both pasture and supplementary feed. During periods of higher pasture growth, excess pasture was cut and ensiled, with two cuts taken from P31 (15 March and 15 November 2018) and one cut from P32 (15 March 2018) during the study period. The harvested supplementary feed along with maize silage produced on the farm and imported supplementary feed (maize silage, pasture silage, hay, straw, and dried distillers grain (DDG)) were fed back to the animals during periods of lower growth. The supplementary feed was initially fed in the grazed paddocks, however, in June 2015 a dedicated feed pad was installed and used for the animals to receive most of the supplementary feed. Supplementary feed usage increased through the study period, providing more feed for the fewer cows and therefore, despite a 38% decrease in total herd size, milk production only decreased by 22%.

Additional nutrients were supplied to the experimental paddocks via application of commercial (synthetic) fertiliser, manures (duck, chicken, and goat – largely composed of animal bedding (wood chip and straw) and excrement), compost, and sludge (solid excrement collected from the feed pad). Commercial fertiliser was applied in all years, with the other forms of nutrient additions only occurring in certain years (Table S1). In most years, P31 and P32 received very similar nutrient quantities except for the 2014 and 2018 years. On average, P31 received 56 kg N ha^{-1} yr^{-1}, 16 kg P ha^{-1} yr^{-1}, and 41 kg K ha^{-1} yr^{-1}, while P32 received 50 kg N ha^{-1} yr^{-1}, 14 kg P ha^{-1} yr^{-1} and 32 kg K ha^{-1} yr^{-1}.

While the management of the farm evolved through the study period, in general, the management of the two experimental paddocks was the same each year. However, there were occasions where the short-term management of the two paddocks differed, primarily through differences in the timing of the grazing of paddocks (referred to as asynchronous grazing). These short-term differences were used to test our methodology and as examples of the information able to be derived by calculating the C balance for each paddock (see below).

2.3. Eddy covariance measurements

2.3.1. Instrumentation

Net ecosystem exchange (NEE) was measured at a location on the boundary between P31 and P32 (Fig. 1). The EC system consisted of a CSAT3 sonic anemometer (Campbell Scientific Inc., Logan, UT, USA (CSI)) and an enclosed path LI-7200 infrared gas analyser (LI-COR Inc., Lincoln, NE, USA; tube length of 0.6 m) located at 1.55 m height. Measurements were made at 20 Hz, by a CR3000 datalogger (CSI), and collected on a compact flash card. Additional supporting soil and meteorological measurements included soil moisture at 50 mm depth (107 probes; CSI); soil temperature using a four-junction averaging thermocouple (TCAV; CSI) with two probes at 20 and 60 mm (referred to as 40 mm); volumetric soil temperature at 50 and 100 mm depth (CS616; CSI); and two soil heat flux plates (HFT01; Hukseflux Thermal Sensors, Delft, Netherlands) at 80 mm depth; air temperature and relative humidity (HMP155; Vaisala, Helsinki, Finland) measured at 1.55 m; shortwave incoming radiation at 1 m (NR01; Hukseflux Thermal Sensors); and rainfall at 450 mm (TB5; Hydrological Services). Photosynthetic photon flux density (PPFD) was used in the calculation of ΔNEE (see below) and was estimated as shortwave incoming radiation divided by 0.48. The supporting soil and meteorological data were averaged (or totalled) every half hour from 1-second measurements, and stored on CR3000 and CR1000 dataloggers (CSI).

2.3.2. Flux processing

Half-hourly CO_2 and H_2O fluxes were computed from the 20 Hz data using EddyPro version 6.2.2 software (EddyPro ® Version 6.2 [Computer software], 2016). Data presented in Rutledge et al. (2017a)

was originally processed using EddyPro version 5.2.0 (EddyPro® Version 5 [Computer software], 2015), and have been reprocessed using modified settings as outlined below. The EddyPro software performed raw data screening for spike removal, amplitude resolution, drop-outs, absolute limits and skewness and kurtosis following Vickers and Mahrt (1997). Further processing settings included the application of double-axis rotation, correction of CO_2/H_2O concentration and vertical wind speed time lags, corrections for high-pass (Moncrieff et al., 2004) and low-pass filtering (Fratini et al., 2012), and correction for sensor separation (Horst and Lenschow, 2009).

Processed flux data were then subjected to further QA/QC, with flux data rejected when certain criteria were not met. Data were rejected during periods of site visits, CO_2 fluxes were removed when instruments (IRGA and CSAT3) either malfunctioned or reported warnings for more than 0.5% of each half-hour period, were unrealistically low ($<$ –40 μmol m^{-2} s^{-1}) or high ($>$ 18 μmol m^{-2} s^{-1}), along with half-hours where Mauder and Foken (2004) quality control flags for stationarity and developed turbulence resulted in a poor quality value of 2. Fluxes collected under low turbulence conditions were rejected using the standard deviation of the vertical wind speed (σ_w; Acevedo et al., 2009), with a σ_w threshold of 0.10 m s^{-1} determined for our site by adapting the methods of Reichstein et al. (2005). Half-hours where the wind originated from behind the supporting tower, or the standard deviation of the wind direction was either too large ($\sigma_{dir} > 40^{\circ}$) or too small ($\sigma_{dir} < 1^{\circ}$) were removed (Hunt et al., 2016). We chose to exclude grazer respiration from our measured fluxes (Kirschbaum et al., 2015), and therefore following footprint analysis (Kormann and Meixner, 2001) any half-hours where more than 5% of the flux footprint was derived from a source area grazed by cows were rejected. Finally, only fluxes derived from within the desired source area according to footprint analysis were retained (see details below).

To calculate annual flux totals for each paddock individually, we additionally filtered the data to include only those fluxes that originated from the desired source area. Half-hourly fluxes that exceeded a threshold value for minimum contribution from the desired source area (i.e. P31, P32 or NewMix) were then assigned to that source area, resulting in three datasets (with the P31 and P32 datasets being subsets of the NewMix dataset). The minimum contribution thresholds chosen were 70% for daytime data and 60% for night-time data. After the QA/QC filtering step, (including assignment of measurements to one paddock or the other) an annual average of 25.9% of data half-hours remained for P31 (32.4% day-time and 19.5% night-time), and 15.7% for P32 (20.1% day-time and 11.4% night-time; Table 1 and Table S2). Slightly less than 50% of NewMix data remained after filtering. The length and distribution of gaps within the P31 and P32 datasets were examined in detail to ensure measurements were attributable to each paddock regularly, and that extended periods (i.e. several days) of no data were minimal. For both P31 and P32 datasets, the majority of gaps were less than 1 day in length (Fig. 2) and where the very few gaps were longer than 7 days (6 occurrences for P31, and 8 for P32), instrument malfunction was the frequent cause thus affecting all datasets. Data gap occurrences for P31 and P32 were also compiled against time of day (Figure S1), weather conditions (Table S2), seasonal variation (Table S2), and management (grazing and harvest) events (Figure S2). While data coverage was not evenly distributed under all scenarios considered, data was consistently available from both P31 and P32, and all conditions represented year-round.

Table 1
Data remaining following QA/QC process and flux contribution from the source area of target dataset for P31 and P32, and NewMix for each year and averaged across all years.

	Data Remaining (%)			Flux Contribution from Source Area (%)		
Year	P31	P32	NewMix	P31	P32	NewMix
2012	24.8	18.6	49.9	81.8	79.5	82.5
2013	24.4	18.1	49.5	82.9	81.3	84.1
2014	26.1	16.0	49.7	82.8	81.3	84.2
2015	25.6	16.5	50.4	82.7	81.2	84.0
2016	29.5	14.4	52.5	83.2	81.2	84.6
2017	24.6	13.9	46.2	83.5	82.0	85.2
2018	26.0	12.5	45.4	84.6	82.8	86.1
Average	25.9	15.7	49.1	83.1	81.4	84.4

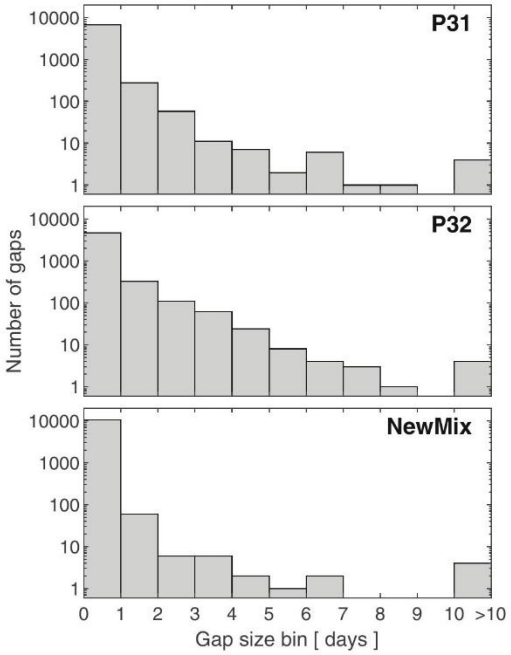

Fig. 2. Gap size distribution for P31, P32 and NewMix NEE datasets for 2012–2018. Each gap size bin includes all gaps which have a length between the days indicated either side of the bar on the x-axis (e.g. gaps which are between 1 and 48 half-hours in length fall into the 0–1 day bin, etc.).

Examination of the flux footprint generated from each dataset confirmed the thresholds chosen were suitable (Fig. 1). Furthermore, >80% of the annual average flux contribution was derived from the target source area for each dataset in all years except P32 in 2012 (Table 1). Each dataset was then gap filled according to the procedure outlined in Section 2.3.3.

2.3.3. Gap-filling procedure

We used an artificial neural network (ANN; Papale and Valentini, 2003) to gap-fill CO_2 and H_2O fluxes, with separate models for daytime and night-time for each flux. ANN settings for gap-filling of CO_2 fluxes followed the methodology of Rutledge et al. (2017a), while Pronger et al. (2016) described the gap-filling of H_2O fluxes. Input drivers for each model (day and night models of CO_2 and H_2O) are summarised in Table 2 and include a combination of air temperature, soil temperature (40 mm and 100 mm), volumetric moisture content (100 mm), ΔNEE (see below), PPFD, vapour pressure deficit, net radiation, and horizontal wind speed. Each ANN model was trained and run 50 times with the resulting gap-filled data being the median prediction of the 50 runs (at the half-hourly time step).

To account for the cyclic removal and regrowth of pasture phytomass by grazing, the phytomass index (PI) was determined from measured NEE. PI is an empirically determined coefficient designed to describe the amount of photosynthetically active vegetation within an ecosystem (Aurela et al., 2001). Campbell et al. (2015) demonstrated that PI was a suitable descriptor of aboveground biomass in grazed

Table 2
Artificial neural network (ANN) input variables for gap-filling CO_2 and H_2O fluxes. Day- and night-time models were run separately. Input variables include: net radiation (R_n), photosynthetic photon flux density (PPFD), vapour pressure deficit (VPD), air temperature (T_a), 40 and 100 mm soil temperature ($T_{S,40}$ & $T_{S,100}$, respectively), 100 mm volumetric moisture content (VMC_{100}), horizontal wind speed (wind speed), and ΔNEE.

Variable	CO_2 Day ANN	CO_2 Night ANN	H_2O Day ANN	H_2O Night ANN
R_N			✓	✓
PPFD	✓			
VPD	✓		✓	✓
T_a	✓	✓	✓	✓
$T_{S,40}$	✓	✓		
$T_{S.100}$		✓		
VMC_{100}	✓	✓	✓	✓
Wind speed			✓	✓
ΔNEE	✓	✓	✓	✓

pastures where the biomass is periodically removed during grazing. Consequently, PI was identified as a useful input variable in the gap-filling process of grazed pastoral systems. As described by Lohila et al. (2004), PI is calculated as the difference between CO_2 fluxes during high-light daytime conditions (PPFD > 700 μmol m^{-2} s^{-1}) and at night (PPFD < 10 μmol m^{-2} s^{-1}) normalised to unity at its maximum, resulting in a daily value between 0 and 1. In this study, we used the non-normalised PI, referred to as ΔNEE, as the input variable into the ANN gap-filling procedure. Using ΔNEE rather than normalised PI resulted in better performance of the ANN in our system (with the same ΔNEE value used for all half-hours of each day). ΔNEE was calculated as average night-time NEE (NEE_{night}) minus the average daytime NEE (NEE_{day}) when the light conditions were above the specified threshold (PPFD > 700 μmol m^{-2} s^{-1} reduced to PPFD > 500 μmol m^{-2} s^{-1} during the winter months of May to August (Campbell et al., 2015)). A minimum of three half-hour NEE values were required to calculate an acceptable NEE_{day} or NEE_{night} value.

Gaps in the NEE flux data resulted in there being times when ΔNEE could not be calculated as less than three (day- or night-time) NEE values were available. Furthermore, partitioning the half-hourly NEE data according to the source area exacerbated the gaps in the calculation of the daily ΔNEE time series for individual paddocks. Before utilising the ANN to fill gaps in the NEE dataset, ΔNEE itself must be gap-filled. In this analysis we calculated, and gap-filled, both NEE_{day} and NEE_{night} separately rather than gap-filling ΔNEE itself.

The gap-filling procedure for NEE_{night} was the same for all datasets (P31, P32, and NewMix). If insufficient night-time NEE data were available to calculate NEE_{night} the averaging period was extended to include the previous and subsequent nights. Any further remaining gaps were filled using linear interpolation. We used different approaches to gap-fill NEE_{day} for the paddocks and NewMix. NEE_{day} is driven by photosynthesis, which is dependent on the amount of aboveground biomass, which in turn is affected by the grazing regime. Therefore, we used the timing of grazing to inform the gap-filling of NEE_{day} for P31 and P32. Gap-filling of NEE_{day} for each paddock was achieved via a two-step process: (1) fitting a linear regression to the available NEE_{day} data as a function of days since the paddock was grazed for each inter-grazing period, and (2) linearly interpolating NEE_{day} for day(s) during grazing (measured or gap-filled NEE_{day} data on the day before and the day after grazing was used as the start- and end-points for the interpolation) (see Figure S3 in supplementary material for an example). The approach described to gap-fill NEE_{day} for individual paddocks was not suitable to gap-fill NEE_{day} for NewMix as grazing of both P31 and P32 rarely occurred on the same day. NEE_{day} for NewMix was gap-filled using linear interpolation.

Performance of the ANN gap-filling for NEE of P31 and P32 was slightly better than for NewMix despite the reduction in the data sets. The relationship between measured and modelled fluxes following gap-filling yielded R^2 values of 0.93, 0.91 and 0.91 for the daytime NEE ANN and 0.63, 0.65 and 0.62 for the night-time ANN (P31, P32, and NewMix respectively). The similar fits of P31 and P32, when compared to NewMix (despite lower data coverage), contrasts with Moffat et al. (2007) who found decreased gap-filling performance with decreased data coverage. This contrast is likely due to the smaller datasets of the individual paddocks containing a single consistent signal, which is not the case for NewMix, which is composed of an integrated signal from both paddocks.

2.4. Net ecosystem carbon balance (NECB)

Net ecosystem carbon balances (NECB) were calculated for each paddock and NewMix annually, by adding all imports and exports of C to the calculated net ecosystem productivity (NEP; equal in magnitude, but opposite in sign to NEE). NECB was calculated as:

$$NECB = NEP + F_{Supp} + F_{manure} + F_{fertiliser} + F_{excreta\ returned} - F_{pasture\ removed} - F_{harvest} - F_{Supp\ removed} - F_{leach} \quad (1)$$

where: *NEP* was net ecosystem production (as described in Section 2.3); F_{Supp} was imported supplemental feed; F_{manure} was added manure (including animal bedding and excrement, compost and sludge); $F_{fertiliser}$ was C added in synthetic fertilisers (predominantly lime and urea); $F_{excreta\ returned}$ was the C deposited on the paddock by the grazing animals as dung and urine; $F_{pasture\ removed}$ was the pasture C removed by the grazing animals; $F_{harvest}$ was pasture C removed by mechanical harvesting; $F_{Supp\ removed}$ the imported supplemental feed C consumed by the grazing animals; and F_{leach} was dissolved organic C leached through the soil below 0.6 m. Emissions of soil CH_4−C are ignored in this study, although they are likely to be very small (e.g. Felber et al., 2015), while CH_4−C emissions from enteric fermentation are accounted for within $F_{pasture\ removed}$. Calculation of all components except $F_{harvest}$ are described in full by Rutledge et al. (2017a), and the reader is referred there for complete details. $F_{harvest}$ was calculated as the total wet mass of pasture as measured by tractor-mounted scales (Alpha Weighlog 10, RDS Systems Ltd.) multiplied by the measured dry matter fraction and carbon content of the pasture. Finally, in this study, the grazing animals were considered external to the system (Felber et al., 2016a; Laubach and Hunt, 2018), and thus NEP excluded grazer respiration and the grazing animals acted simply as transporters of pasture, supplementary feed, and excreta.

2.5. Uncertainty estimates

In this study, we report uncertainties as 95% probability intervals. This method used an approach inspired by Bayesian elicitation (O'Hagan et al., 2006), and firstly involved determination of probability distributions (assumed Gaussian) for all data used in the NECB calculations. From the probability distribution, 1000 estimates of each value were drawn, followed by calculation of all components of the NECB producing 1000 estimates of each. Finally, the 95% probability intervals were calculated from the 2.5^{th} and 97.5^{th} percentiles of the 1000 estimates, with differences considered significant if probability intervals did not overlap.

Probability distributions for non-NEP components were produced from a combination of measurements and estimates. Where available, measurements of several samples (e.g. the dry matter content of harvested biomass) allowed for the calculation of a mean and variance to determine the probability distribution. However, where data resulted from a single measurement or estimate, a different approach was used. Here, the available value was assumed to be the mean, and variance was estimated from 'expert opinion' allowing a probability distribution to be simulated. Further details for the calculation of the non-NEP component uncertainty are described in Rutledge et al. (2017a).

For NEP, the variance (and thus probability distribution) was estimated using established methods to account for both random and systematic sources. In this study four components of NEP uncertainty were considered and computed (as standard deviations) separately:

(1) Random error (σ_r) associated with both the measured and gap-filled data (separate from uncertainty due to the gap-filling procedure itself) was calculated using the procedure described by Dragoni et al. (2007).
(2) Uncertainty arising from the gap-filling procedure (σ_{GF}) was estimated by evaluating the 50 runs of the ANN used to produce the gap-filled data. Each ANN run was trained on a different data subset (70% of the initial dataset) and therefore providing 50 differing estimates of each modelled flux. On occasion, the ANN produced completely implausible values (e.g. night-time NEE of ~100 μmol m^{-2} s^{-1}), and therefore the ANN runs were first screened for plausibility (–40 > NEE < 18 μmol m^{-2} s^{-1}). Any ANN runs containing non-plausible values were discarded and resulted in 5% of P31 runs and 7% of P32 runs being excluded (no runs for NewMix were discarded). Measured, filtered values were inserted into the ANN simulations, and annual totals calculated. The uncertainty was computed as the standard deviation of the annual totals.
(3) To assess the uncertainty associated with the choice of σ_w threshold in the filtering step (σ_{turb}) we first calculated annual NEP totals using a range of plausible σ_w thresholds near our chosen default (Elbers et al., 2011; Hunt et al., 2016). These thresholds were ± 0.02 and ± 0.04 m s^{-1} from our default (0.10 m s^{-1}). We then calculated σ_{turb} as the standard deviation of the resulting five annual NEP datasets.
(4) The final uncertainty considered was that of choice of threshold for the minimum source area contribution from footprint analysis (σ_{FP}). We used the same approach as for determining σ_{turb} by varying the minimum contribution threshold and calculating the standard deviation of the totals. The default thresholds of 70% day-time and 60% night-time contribution were varied by ± 5% and ± 10% resulting in five sets of annual totals.

The uncertainty for NEP (σ_{NEP}) was subsequently calculated as:

$$\sigma_{NEP} = \sqrt{\sigma_r^2 + \sigma_{GF}^2 + \sigma_{turb}^2 + \sigma_{FP}^2}$$

For NewMix, all sources of uncertainty contributed relatively evenly to the σ_{NEP}, while for P31 and P32 σ_{GF} was the largest contributor likely due to the reduced sizes of the data sets compared to NewMix. The calculated uncertainties of P31, P32 and NewMix varied considerably from year-to-year, predominantly due to variation in σ_{FP}.

3. Results

3.1. Annual fluxes

3.1.1. NEP

Annual NEP for P31, P32, and NewMix were positive in all years, ranging from 108 g C m^{-2} y^{-1} (P31 in 2017) to 436 g C m^{-2} y^{-1} (P32 in 2016) (Fig. 3). P32 annual NEP were more variable (range 156–436 g C m^{-2} y^{-1}) than P31 (108–345 g C m^{-2} y^{-1}), although when uncertainties were considered only 2014 and 2016 years had significantly different CO_2 uptake. Four of the seven measurement years resulted in differences between the paddocks of >50 g C m^{-2} y^{-1}, with the 2016 annual difference of 204 g C m^{-2} y^{-1} being the largest. Despite the inter-annual variability between the two paddocks, after seven years, the mean annual NEP (± 95% confidence interval) for P31 was 263 ± 58 g C m^{-2} y^{-1}, while P32 was 284 ± 61 g C m^{-2} y^{-1}, and the two paddocks were not significantly different from each other (Table 3).

The NewMix annual NEP was between the totals of P31 and P32 for four of the seven years (Fig. 3a). NewMix annual NEP varied from 152 g C m^{-2} y^{-1} (2017) to 392 g C m^{-2} y^{-1} (2012), and the seven-year average was 256 ± 61 g C m^{-2} y^{-1} (Table 3). Such a result is unsurprising given the NewMix annual totals are calculated from data derived from both P31 and P32. Accordingly, for comparative purposes, a composite annual flux ($NewMix_{Composite}$) was calculated from the individual paddock annual fluxes weighted by the annual flux contribution originating from that paddock. $NewMix_{Composite}$ only differed significantly from NewMix for the 2013 year (Fig. 3a). A significant difference between NewMix and $NewMix_{Composite}$ may suggest NewMix was either over- or underestimated, which can occur due to the changing source footprints coupled with differences in grazing status (see below).

An example of the overestimation of NewMix NEP relative to P31 and P32 (and thus $NewMix_{Composite}$) was observed between 27 October and 14 November 2017 (Fig. 4). Grazing of P31 occurred on 26 October and P32 on 27 October, with the next grazing of both paddocks on 14 November. Daily NEP for P31 decreased to about –2 g C m^{-2} d^{-1} following grazing on 26 October before slowly increasing to about 2 g C m^{-2} d^{-1} through to the next grazing on 14 November (Fig. 4b). In contrast, grazing of P32 on 27 October resulted in daily NEP reducing to about –3 g C m^{-2} d^{-1} and remaining of similar magnitude for several days until 5 November, after which daily NEP gradually increased to about 4 g C m^{-2} d^{-1} prior to the subsequent grazing on 14 November. For the first half of this example period, the source area for the measured flux was predominantly P31 (Fig. 4a), and thus the NewMix daily flux closely followed that of P31 (Fig. 4b) and was higher than for P32. During the second half of the period, P32 became the more dominant source area, and therefore the NewMix flux more closely aligned with the flux of P32, which was generally higher than that of P31. When the daily fluxes for the period were accumulated (Fig. 4c), total NEP for paddock's 31 and 32 (0 and –6 g C m^{-2}, respectively) were lower than for NewMix (17 g C m^{-2}). In this example, the changing source area coupled with differing patterns of CO_2 uptake/release for P31 and P32 resulted in NewMix NEP exceeding that of either individual paddock.

3.1.2. Evaporation

For comparative purposes, we also considered evaporation rates (E). Annual evaporation of P32 (range 694–767 mm y^{-1}) was greater than P31 (range 664–767 mm y^{-1}) by between 10 and 32 mm y^{-1} in all years except 2012 when they were both 767 mm. NewMix annual evaporation (669 to 776 mm y^{-1}) was between that of P31 and P32 in all years except 2012 when it was slightly greater than either P31 or P32 (Fig. 3c). Calculation of $NewMix_{Composite}$ for E resulted in a maximum annual difference between $NewMix_{Composite}$ and NewMix of less than 9 mm y^{-1}, and indeed for most years were within 4 mm y^{-1}.

3.2. Annual C balances

While eddy covariance determined fluxes from grazed pastoral systems have formed the basis of individual studies (e.g. Felber et al., 2016b; Gourlez de la Motte et al., 2018), when combined with other flows of C allows for the calculation of full C balances (NECB). In the following, we report the NECB for each of the two paddocks and NewMix.

Like annual NEP, significant differences in annual NECB between P31 and P32 were observed in two of the measurement years (Fig. 3b). NECB for P31 ranged from –150 g C m^{-2} y^{-1} (2017) to 172 g C m^{-2} y^{-1} (2014), while P32 ranged from –107 g C m^{-2} y^{-1} (2017) to 200 g C m^{-2} y^{-1} (2016) (Table 3). During the study period, the paddocks were significant sinks of C, significant sources of C or C neutral, and often changed from one to the other in sequential years. Despite the year-to-year variability, the seven-year average NECB's were 1 ± 76 g C m^{-2} y^{-1} and 7 ± 80 g C m^{-2} y^{-1} for P31 and P32 and were not significantly different from each other. Therefore, long

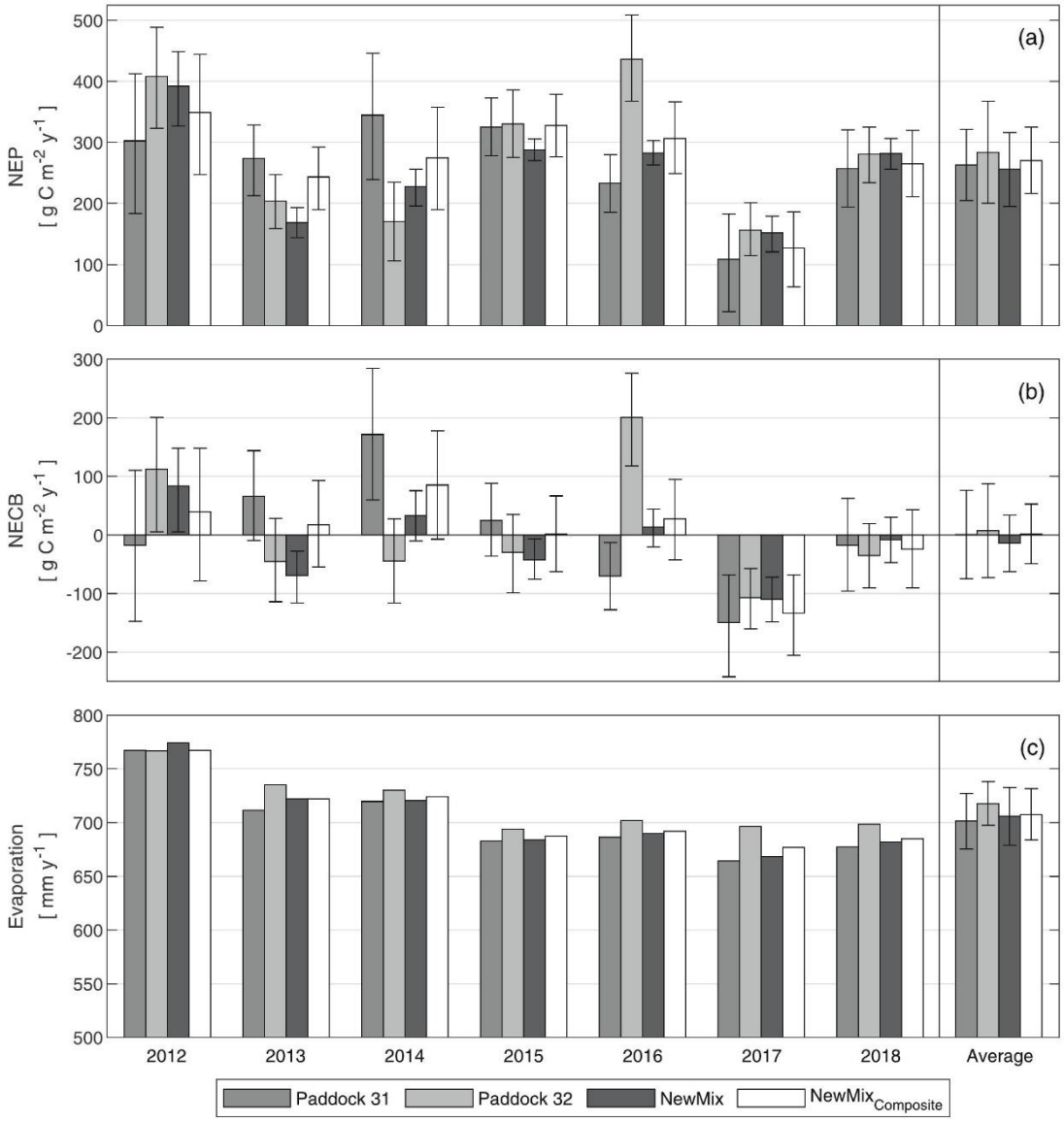

Fig. 3. NEP (a), NECB (b) and evaporation (c) annual totals for each measurement year, and an average for all years calculated for P31, P32, NewMix and $NewMix_{Composite}$ (calculated as annual P31 and P32 fluxes weighted by the footprint contribution from that paddock). Errors bars for the individual years in (a) and (b) represent 95% probability intervals. Uncertainties of annual averages are 95% confidence intervals of the mean.

term, both paddocks were C neutral.

The patterns of NewMix annual NECB relative to P31 and P32 mirrored that of NEP and NewMix NECB was between that of P31 and P32 for the same four measurement years as NEP (Fig. 3b). The NewMix NECB for 2013 and 2015 was lower than both P31 and P32, and for 2018 NewMix NECB was higher. For the 2013 year, NewMix was significantly different from P31, but not from P32, while for 2016 NewMix was significantly different from P32 but not P31. The seven-year average NECB for NewMix was –14 ± 48 g C m^{-2} y^{-1} and was lower than both P31 and P32 but not significantly different from either. Like P31 and P32, the seven-year average NECB for NewMix was C neutral.

3.3. Effect of specific management practices

3.3.1. Impact of asynchronous grazing

Management of grasslands employing rotational grazing can result in significant biomass removal in a period as short as a few hours resulting in rapid changes in the photosynthetic capacity of the pasture. For instance, the average stocking density during grazing was 187 cows ha^{-1}, and average grazing duration was 25 h (range 4.5 h to 4.5 days) during which time ~1000–1500 kg DM ha^{-1} was removed. The effect of this rapid biomass removal is often observed in the measured CO_2 flux with daily total NEP being reduced for several days following grazing relative to immediately before grazing. Clear examples of the effect grazing has on NEP can be seen in Fig. 5. Asynchronous grazing of paddocks can lead to each paddock having differing NEP regimes, and

Table 3
Annual NECB and components for P31, P32, and NewMix for all measurement years. The seven-year average and 95% confidence interval of the mean are also reported. Units are g C m^{-2} y^{-1}.

	NEP	F_{Supp}	F_{Manure}	$F_{Fertiliser}$	$F_{Pasture\ Removed}$	$F_{Harvest}$	$F_{Supp\ Removed}$	F_{Dung}	F_{Urine}	F_{leach}	*NECB*
Paddock 31											
2012	302	37	64	0	−625	0	−29	206	28	−2	−18
2013	274	57	111	1	−473	0	−45	126	17	−2	66
2014	345	77	152	11	−527	0	−62	157	22	−2	172
2015	325	43	10	1	−505	0	−34	164	23	−2	24
2016	233	50	0	11	−563	0	−40	212	29	−2	−70
2017	108	33	0	7	−465	0	−26	174	24	−4	−150
2018	257	25	44	2	−342	−131	−20	132	18	−3	−17
Average	*263*	*46*	*54*	*5*	*−500*	*−19*	*−37*	*167*	*23*	*−2*	*1*
95% CI	*58*	13	*44*	*4*	*66*	*37*	*10*	*25*	*3*	*1*	*76*
Paddock 32											
2012	408	35	96	0	−616	0	−28	193	27	−2	112
2013	204	76	111	1	−548	0	−61	153	21	−2	−45
2014	170	58	93	11	−479	0	−46	134	18	−2	−44
2015	330	41	8	1	−546	0	−33	150	20	−2	−30
2016	436	47	0	11	−520	0	−38	233	32	−2	200
2017	156	12	0	7	−439	0	−10	150	21	−4	−107
2018	281	18	4	2	−416	−92	−14	163	22	−3	−35
Average	*284*	*41*	*45*	*5*	*−509*	*−13*	*−33*	*168*	*23*	*−2*	*7*
95% CI	*84*	*16*	*39*	*4*	*51*	*26*	*13*	*25*	*4*	*1*	*80*
NewMix											
2012	392	36	80	0	−620	0	−29	199	27	−2	84
2013	168	66	111	1	−521	0	−53	141	19	−2	−69
2014	228	67	122	11	−503	0	−54	145	20	−2	33
2015	288	42	9	1	−525	0	−33	157	22	−2	−42
2016	282	49	0	11	−541	0	−39	223	31	−2	14
2017	152	22	0	7	−452	0	−18	162	22	−4	−110
2018	281	21	24	2	−374	−111	−17	148	20	−3	−9
Average	*256*	*43*	*49*	*5*	*−505*	*−16*	*−35*	*168*	*23*	*−3*	*−14*
95% CI	*61*	*14*	*40*	*4*	*57*	*31*	*11*	*23*	*3*	*1*	*48*

therefore the NewMix NEP depended on the flux source location. The effect of grazing on the NEP regime of an individual paddock is clear, however, this can become smoothed, or missed entirely when NEP is aggregated across both paddocks with changing source areas. For example, Fig. 5 illustrates a clear grazing signal from P32 on 23 September 2013, which was not observed in the NewMix signal. Similarly, the effect of increased NEP prior to the grazing of P32 on 17 October and associated decreased NEP following grazing is muted by the influence of P31, which was midway through a grazing cycle.

3.3.2. Effect of heavy grazing leading to pugging

Calculating CO_2 fluxes for individual paddocks rather than treating multiple paddocks as a single site offers the potential to derive information otherwise unavailable. During the 2017 winter, P31 was heavily grazed over 4–7 July 2017 (Fig. 6a). In the three days prior to the beginning of this grazing event, there had been 63 mm of rainfall, with a further 34 mm during the grazing event so that water-filled pore space (WFPS) increased from 77% to 92% (Fig. 6b). Consequently, a large proportion of P31 became heavily pugged (defined as deformation and remoulding of soil when animal loads exceed a soil's bearing capacity (Houlbrooke et al., 2009)). P32 was also grazed on 7 and 8 July, but the grazing duration was only 13 h per day (compared to 20 h per day for P31), and the daily stocking rate was much lower (c.f. 67 vs 170 cows ha^{-1}). The reduced grazing duration and stocking rate resulted in negligible pugging of P32. This combination of events and circumstances, along with the ability to calculate fluxes for each paddock individually allowed us to estimate the potential impact the heavy grazing and thus pugging had on NEP (Fig. 6c) and therefore NECB (due to the absence of other flows of C). Flux data patterns for both paddocks in the two weeks before the grazing events were almost identical, thus the two paddocks were considered to be following a similar short-term CO_2 exchange trajectory. However, following the grazing and pugging of P31, the daily CO_2 (net) losses during the next 2–3 weeks from P31 were more than twice that of P32. At the time of next grazing on 25 August (both paddocks), accumulated NEP (since the conclusion of P32 grazing on 8 July) for P31 was 6 g C m^{-2} and 38 g C m^{-2} for P32. The difference in cumulative NEP between P31 and P32 of 32 g C m^{-2} provides an example of the magnitude of the potential effect a heavy grazing event in wet conditions resulting in pugging can have on NECB.

4. Discussion

In general, our results demonstrate that it is feasible to calculate annual NEP and NECB totals for two individual paddocks adjacent to a single EC system, and highlighted benefits of calculating flux totals at the paddock scale. The seven-year averages (and 95% confidence intervals) for NEP were 263 ± 58 g C m^{-2} y^{-1}, 284 ± 84 g C m^{-2} y^{-1} and 256 ± 61 g C m^{-2} y^{-1} (P31, P32 and NewMix, respectively). For NECB these were 1 ± 76 g C m^{-2} y^{-1}, 7 ± 80 g C m^{-2} y^{-1} and –14 ± 48 g C m^{-2} y^{-1} (again for P31, P32, and NewMix, respectively).

4.1. Paddock scale fluxes

Prior to the assessment of the paddock scale fluxes, it is worthwhile briefly considering the effect of partitioning the measured half-hourly measurements had on data coverage. Assignment of half-hourly flux data to individual paddocks reduced average data availability from 49.1% (total data coverage) to 25.9% for P31 and 15.7% for P32 (Table 1). This level of data coverage was lower than typically reported for grassland EC studies, however, especially for P31, did approach similar levels to other studies where annual CO_2 balances have been reported. For example, Ammann et al. (2007), Campbell et al. (2014) and Eichelmann et al. (2016) all report data coverages of between 30% and 35% for multiple site years. Furthermore, Campbell et al. (2014) report a night-time data coverage of 16.6%, which was not much higher than that for P32 (11.4%) in the present study.

Despite the relatively low data coverage, annual NEP values for P31

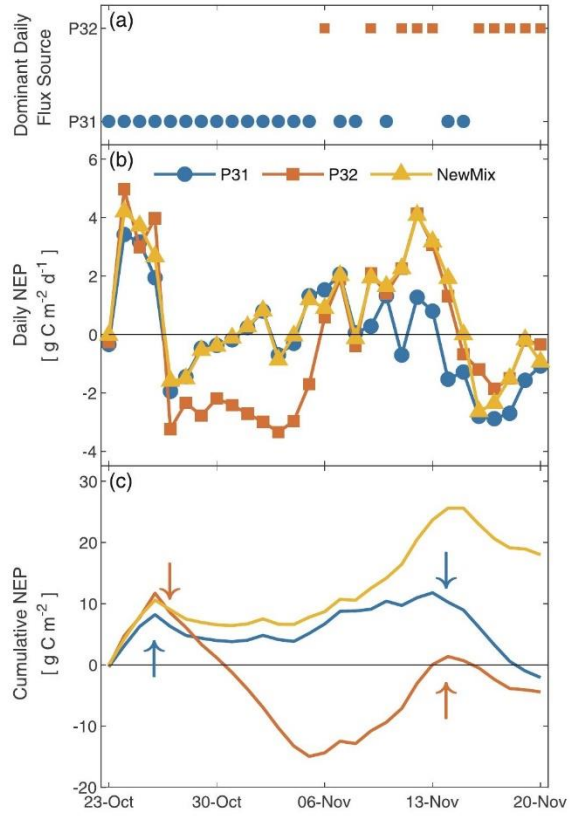

Fig. 4. Example period (23 October to 20 November 2017) illustrating how NewMix NEP can exceed NEP for either P31 or P32. (a) identifies the dominant source paddock for each day based on the calculated footprint; (b) daily total NEP for P31, P32, and NewMix; and (c) cumulative NEP for P31, P32, and NewMix during the example period. In (c) the arrows indicate the start of grazing events for P31 (blue arrows) and P32 (red arrows). Positive values represent a net uptake and negative values a net release of CO_2. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

and P32 were within the range reported from other managed grassland studies globally, although towards the higher end (e.g. see Fig. 4 in Rutledge et al., 2015). However, when compared to other New Zealand studies (Campbell et al., 2015; Rutledge et al., 2015; Laubach and Hunt, 2018; Wall et al., 2019) average annual NEP (263 ± 58 g C m^{-2} y^{-1} for P31 and 284 ± 84 g C m^{-2} y^{-1} P32) were similar. At a nearby dairy farm Wall et al. (2019) reported a very similar three-year mean NEP of 258 ± 114 g C m^{-2} y^{-1} for the June 2013 to May 2016 period. An earlier study on a farm ~40 km away by Rutledge et al. (2015) measured a four-year average NEP of 165 ± 71 g C m^{-2} y^{-1}, although this study included grazer respiration in calculation of NEP which has been identified as problematic (Kirschbaum et al., 2015), and can significantly reduce the reported NEP (Gourlez de la Motte et al., 2019). Additionally, the highest annual NEP of 436 g C m^{-2} y^{-1} observed here (P32, 2016) was similar to that reported by Laubach and Hunt (2018) from a rotationally grazed and irrigated NZ pasture. While annual NEP calculated for P31 and P32 varied considerably both inter-annually, and between paddocks, their magnitudes were in good agreement with expected values based on local and international literature.

Although the individual paddock fluxes were often different from each other, confidence in the calculation of the annual totals was derived from two lines of evidence. Firstly, annual NEP integrated across both paddocks (NewMix), and the composite NEP ($NewMix_{Composite}$) calculated from the paddock fluxes and the weighted annual footprint contributions were generally in reasonable agreement. Except for the 2013 year, NewMix and $NewMix_{Composite}$ were within 47 g C m^{-2} y^{-1} of each other. Even in the year with the largest difference in annual NEP between P31 and P32 (203 g C m^{-2} y^{-1}; 2016), the difference between NewMix and $NewMix_{Composite}$ was only 24 g C m^{-2} y^{-1}. However, it would not necessarily be expected that the NewMix and $NewMix_{Composite}$ annual NEP totals should completely agree. Due to the impacts of low-level management decisions (e.g. grazing timing, duration, and stocking rates), short-term fluxes from the two paddocks were at times different (e.g. Figs. 5 and 6), which when coupled with the changing nature of the dominant source area, had potential to lead to an over- or under-estimation of NewMix annual NEP (e.g. Fig. 4).

Unlike NEP, evaporation was not influenced by the low-level management decisions, but rather primarily by available energy (Pronger et al., 2016), and therefore provides an independent assessment of the methods used in this study. A pattern of higher annual evaporation (*E*) from P32 than P31 was observed in all years except 2012, and because the two paddocks were adjacent, the effect of spatial variation on available energy was likely negligible. While the annual *E* of P31 and P32 suggested the paddocks may inherently differ, the difference was <5% in all years, and although not calculated in this study, was therefore likely within the range of uncertainty (e.g. 5.9% in Laubach et al. (2019); 8.8% in Ryu et al. (2008)). Calculation of $NewMix_{Composite}$ *E* and comparison with NewMix *E* showed annual differences of <9 mm y^{-1} (and mostly <4 mm y^{-1}; equivalent to <1% and <0.5% of annual *E* respectively) and thus providing assurance that the method used in this study to calculate the individual paddock fluxes was valid.

4.2. Variability of NECB

4.2.1. Inter-Annual variation

The range of calculated annual NECB's (–150 to 200 g C m^{-2} y^{-1}), although large, was similar to those reported in other managed grassland studies (e.g. Soussana et al., 2007; Gourlez de la Motte et al., 2016; Jones et al., 2017). Of more interest though is the large inter-annual variability. Indeed, the year-to-year differences in NECB for a given paddock ranged from almost no difference (P32, 2013 and 2014) to >300 g C m^{-2} y^{-1} (P32, 2016 and 2017). While not as extreme, P31 had year-to-year differences up to 148 g C $m^{-2}y^{-1}$, and NewMix up to 153 g C m^{-2} y^{-1}. This large inter-annual variability of managed grasslands is not unique, with multiple year studies reporting NECB's with inter-annual differences of up to 181 g C m^{-2} y^{-1} (Zeeman et al., 2010) and 504 g C m^{-2} y^{-1} (Jones et al., 2017). Even multiple-year studies where annual NECB's were more constrained (e.g. Gourlez de la Motte et al., 2016; Wall et al., 2019) inter-annual variability approached 100 g C m^{-2} y^{-1}. Large interannual variability of C balances, as observed in this and other studies, suggests the interpretation of NECB's from individual years needs to be very carefully considered, especially if used as a pre-treatment control year (e.g. Rutledge et al., 2017a).

4.2.2. Between paddock variation

There was also large variability in NECB between P31 and P32 for any given year. These differences in annual NECB between paddocks were mainly a result of differences in NEP. Summation of all the non-NEP terms of the NECB show annual differences between P31 and P32 ranging from 5 to 67 g C m^{-2} y^{-1} (Table 3). For comparison, annual NEP varied between the paddocks by 5 to 203 g C m^{-2} y^{-1}. The similarity between the two paddocks in the non-NEP components was not surprising as the high-level management (e.g. annual number of grazings, fertiliser application, etc.) of both paddocks was generally the

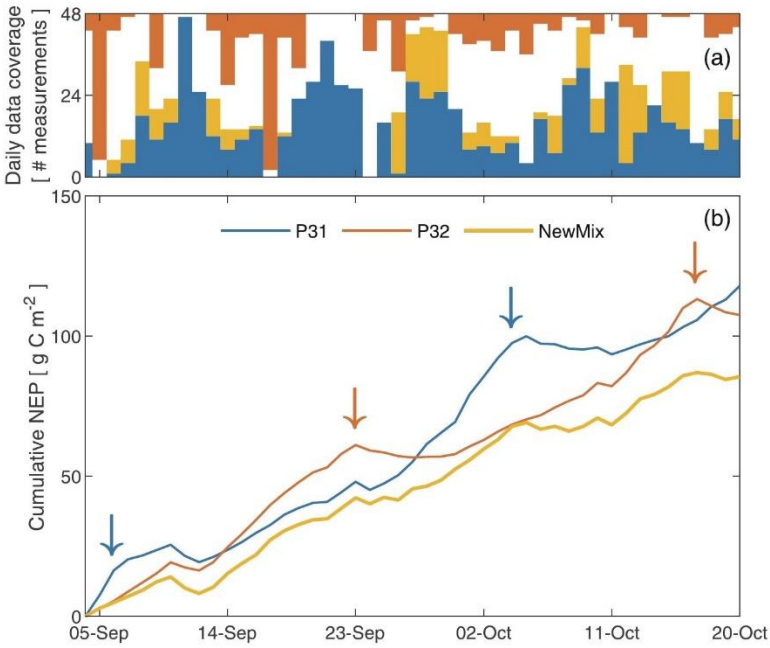

Fig. 5. Cumulative NEP and data coverage for an asynchronous grazing example period in 2013. The top plot (a) indicates the daily data coverage (represented as the number of half-hourly measurements) from each paddock. The blue (lower) bars indicate fluxes sourced from P31; the red (upper) bars indicate fluxes sourced from P32; the gold (central) bars indicate additional valid fluxes where neither P31 or P32 was the dominant source area (and thus contributed to the NewMix dataset); and the remaining white space indicate fluxes removed during the QA/QC process. The bottom plot (b) displays cumulative NEP during the example period, with the arrows indicating the beginning of a grazing event for P31 (blue arrows) and P32 (red arrows). (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.).

same. After seven years, P31 had been grazed or harvested 79 times compared to 78 events for P32 (average 11.3 and 11.1 events per year respectively), and the number of events for each paddock for each year was never more than one event different (the effect of which is quantified below). Furthermore, for all non-NEP components, the difference between P31 and P32 in the annual average for each component was less than 10 g C m^{-2} y^{-1}. This confirms a lack of systematic management differences between the paddocks.

Although P31 and P32 were managed the same in the general sense, the grazing of the paddocks was often asynchronous (e.g. Fig. 5). The

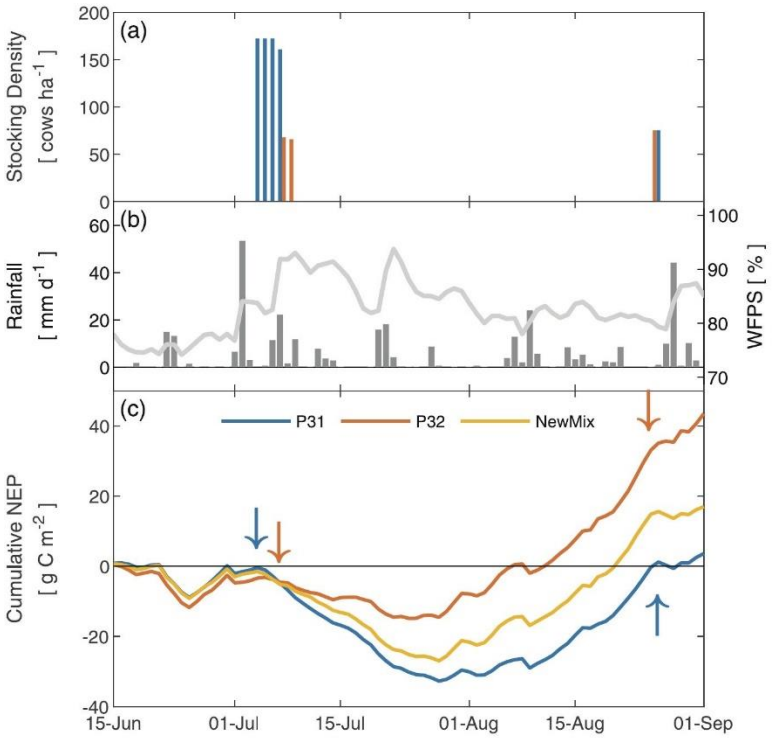

Fig. 6. Example period illustrating the use of adjacent paddock fluxes to identify the impact a heavy grazing event (of P31) in wet conditions that resulted pugging of the soil had on NEP. Panel (a) shows the mean daily stocking rate for P31 (blue bars) and P32 (red bars); panel (b) plots the total daily rainfall (dark grey bars) and mean daily water-filled pore space (WFPS) at 5 cm (grey line); and (c) cumulative NEP for the 15 June to 1 September 2017 period, with the blue arrows indicating the start of grazing events of P31, and the red arrows the start of grazing events of P32. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

importance of differences in the timing of grazing on NECB is particularly relevant with regard to the start and end dates of the time period of interest. For example, if the start (or end) date of a measurement period falls after one paddock has been grazed, but the second has not (and potentially the paddocks could be grazed on consecutive days), the reported C balance can be quite different even if the paddocks are in fact identical (from a C gain/loss perspective). In our system, grazing events typically remove 1000–1500 kg DM ha^{-1} (equivalent to ~45–67.5 g C m^{-2}), and thus in the hypothetical scenario above the C balance of the paddock grazed after the start date could have a C balance of 30–45 g C m^{-2} lower (assuming ~⅓ of the eaten pasture C is returned as excreta) than the paddock grazed before start date. Indeed, such a scenario did occur during our study: for the 2013 year, P32 was grazed on 30 Dec 2013, while P31 was grazed 4 days later on 3 Jan 2014. For the 2013 calendar year (i.e. 1 Jan to 31 Dec) the NECB's of P31 and P32 were 66 and –45 g C m^{-2} y^{-1}, with P32 experiencing one extra grazing event than P31 (i.e. that of 30 December 2013). However, had the start and end dates of measurement year been adjusted to include an equivalent number of grazings and finish just prior to the 30 Dec 2013 grazing of P32 (i.e. the year extending from 30 Dec 2012 to 29 Dec 2013), the NECB's would have been 57 and –15 g C m^{-2} y^{-1} for P31 and P32 respectively. The difference in NECB for P32 (–45 to –15 g C m^{-2} y^{-1}) was attributable to the net effects of grazing (pasture removal and excreta deposition). In contrast, P31's NECB in this example decreased (66 to 57 g C m^{-2} y^{-1}) because the 1 Jan to 31 Dec year included the high net daily uptake of CO_2 observed on 30 and 31 Dec 2013 (5.3 g C m^{-2} d^{-1}) resulting from the extensive biomass present just before grazing. Daily NEP for P31 on 30 and 31 Dec 2012 (as included in the 30 Dec to 29 Dec year) was only 0.8 g C m^{-2} d^{-1} as grazing had occurred only a few days earlier on 20 Dec 2012, and thus photosynthetic uptake was relatively lower. To summarise, care needs to be taken when interpreting temporal C balances from rotationally grazed pastoral systems as the timing of grazing (or harvest) events can have a significant impact on the calculated NECB. Furthermore, where studies compare paddocks (e.g. this study) or indeed sites (e.g. Rutledge et al., 2017a) the implications of grazing timing need to be carefully considered, especially at shorter time scales (e.g. annually or shorter).

Other management decisions such as daily stocking density and grazing duration can also affect paddock-to-paddock variability of C balances as illustrated by the heavy grazing event in wet conditions case study described in Section 3.3.2. In this instance, P31 and P32 were grazed at a similar time, but P31 was grazed at a higher stocking rate, and for a longer duration (Fig. 6). The result was a difference in NEP between the paddocks of 32 g C m^{-2} for the inter-grazing period. We attribute this difference to a reduction in biomass production reducing photosynthetic inputs, and thus NEP. Although we measured biomass, the measurements were insufficient to confirm this hypothesis, however, elsewhere pugging has been shown to decrease pasture growth (Menneer et al., 2005; Drewry et al., 2008; Houlbrooke et al., 2009).

Reporting NECB across an area consisting of multiple paddocks has the effect of smoothing any paddock-to-paddock variation arising due to management effects or otherwise. For example, in our study, the two paddocks were the same size, so the non-NEP components of NewMix were the average of P31 and P32. Furthermore, the impact asynchronous grazing had on NEP was reduced (Fig. 5). Additionally, the annual NECB's become more constrained (i.e. the NewMix range of NECB's was 194 g C m^{-2} y^{-1}, while for P31 and P32 this was 322 and 307 g C m^{-2} y^{-1} respectively), and the uncertainty of the annual mean NECB reduced (± 76, ± 80 and ± 48 g C m^{-2} y^{-1} for P31, P32 and NewMix respectively). It should, however, be noted that the previous study by Rutledge et al. (2017a) identified a large difference in annual NECB of 160 g C m^{-2} y^{-1} between separate EC systems/sites. Consequently, the apparent benefit of integrating across multiple paddocks (smoothing the effect of individual paddock management) may not translate to an increased comparative ability relative to the adjacent paddocks/single EC system method of our study. Calculation of a carbon balance across multiple paddocks may be advantageous for certain studies (e.g. treatment comparison) if management and response are not identical, however, it may be disadvantageous if the investigation requires interpretation of controlling factors, especially if management is asynchronous.

4.3. Methodological considerations

The methodology we tested was suitable for our study site, however, application at other field sites may not be suitable. Firstly, any prospective study site needs to be scrutinised to ensure flux data will be derived from all source areas to avoid biases associated with, for example, changing seasonal wind direction. Here, we had the benefit of using existing data and thus had the opportunity to examine flux footprint data before beginning analysis. Establishment of a new site does not always afford this opportunity, but an assessment of seasonal wind roses coupled with the orientation and layout of the prospective paddocks/fields of interest should provide sufficient information to determine the suitability of the site for such experimental design.

While we used the Kormann and Meixner (2001) footprint model to assign fluxes to each paddock, other methods could be used (e.g. Kljun et al., 2015), or perhaps simple filtering based on wind direction. Of most importance is ensuring that the fluxes used are primarily derived from the source area of interest, and give a true signal of the flux originating from that area.

The assignment of fluxes to different source areas has the consequential effect of reducing data availability for each of the source areas. Therefore, the choice of the infrared gas analyser (open or (en) closed-path) needs to be given consideration. For example, EC studies in New Zealand have used a combination of open and (en)closed path IRGA's, with sites utilising (en)closed path IRGA is generally having higher data coverage. Two studies using enclosed path IRGA's reported 50% (Pronger et al., 2016) and 60% (Hunt et al., 2016) data coverage, while studies using open-path IRGA's had data coverage ranging from 32% to 51% (Campbell et al., 2014; Rutledge et al., 2015; Ratcliffe et al., 2019; Wall et al., 2019). Therefore, selection of an (en) closed path IRGA (as in this study) in EC studies utilising this methodology may boost data coverage to sufficient levels to aid calculation of annual flux totals.

Because our method results in temporal datasets that can be (relatively) small, the use of complementary chamber based measurements may also be beneficial. Direct comparison of the chamber and EC methods has shown agreement to vary from poor to good under different conditions and methodologies (e.g. Riederer et al., 2014; Lucas-Moffat et al., 2018) and, accordingly, Eugster and Merbold (2015) suggested the one does not necessarily provide validation of the other. Nonetheless, chamber based flux data may provide supporting evidence, for example, by allowing for direct comparison of measurements, albeit at a much smaller spatial scale.

Finally, our approach of taking a subset of flux measurements to gap-fill and create a continuous time series worked well for CO_2 and H_2O fluxes, but caution must be advised for other gases (e.g. CH_4, N_2O, etc.). Gap-filling routines for CO_2 and H_2O exchange, both in general (see: Moffat et al., 2007) and for grazed pastoral systems (e.g. Campbell et al., 2015), are well developed, however, this is not yet the case for other gases (Nemitz et al., 2018).

4.4. Application

Measurement of EC fluxes, and indeed calculation of NECB, from grazed grassland systems, is best undertaken on areal extents that are limited to a single paddock and thus reduce the impact of management variation across multiple paddocks. Where this is not possible due to constraints in paddock size relative to the required measurement area

(such as this study), locating the EC system on the boundary between paddocks, and thus allowing the potential for calculation of fluxes/C balances for each paddock, is advisable. It must be noted that expansion to differentiate fluxes to more than two adjacent paddocks may be implausible due to either an inability to distinctly attribute the flux to an individual paddock or insufficient data coverage.

Experimental designs where flux results are reported from two adjacent paddocks using a single EC system has several beneficial opportunities. Primarily, there is the opportunity for replication at a spatial scale much larger than is possible with other techniques (e.g. chamber studies), a key limitation of many EC related studies (Hill et al., 2017). We reported the C balance for two adjacent paddocks under the same general management, however, the methodology would also be suitable for treatment experiments where each paddock had some different management imposed. Various studies have compared different forms of management/treatments using multiple EC systems (e.g. Allard et al., 2007; Ammann et al., 2007; Skinner, 2008) where application of our methodology would have had the potential to yield greater statistical power (i.e. $n > 1$) for the same resource outlay.

While we make the case that our methodology provides the opportunity for replication, care must be taken with experimental design, particularly in rotationally grazed systems. After seven measurement years, we determined the two adjacent paddocks to have the same long term C balance (averaging 1 and 7 g C m^{-2} y^{-1} for P31 and P32 respectively), however, considering individual years in isolation may not have given the same conclusion. Both paddock-to-paddock individual year, and single paddock inter-annual variation was large, likely because of asynchronous grazing and management, coupled with arbitrary start and end dates. However, increasing the number of measurement years reduced the mean annual difference between the paddocks (Fig. 7). For our study site, Fig. 7 suggests that the observed difference between P31 and P32 in both NEP and NECB following seven years of measurements could have been identified using any five years. A shorter period, however, may have concluded the difference to be larger. The ramifications of this finding indicate multiple years are required to conclude similarity or difference between two sites at the annual timescale (whether the interest is in flux totals or C balances). Expanding this concept further, if testing different treatments, the size of an annual difference that could be detected would be expected to decrease as measurement time increased. For example, our data would suggest that after one year of measurement, any difference between paddocks of less than 185 g C m^{-2} y^{-1} (upper bound of a 95% confidence interval) could be due to between paddock variability, however after five measurement years a difference of about 20 g C m^{-2} y^{-1} may be able to be detected. This conclusion also likely applies whether the study compares individual paddocks (e.g. Allard et al., 2007; Ammann et al., 2007) or sites each comprising of multiple paddocks (e.g. Skinner, 2008; Rutledge et al., 2017a). While we believe the above statement to be true for longer periods (such as annually), it is also possible to make valid short-term comparisons between adjacent paddocks, particularly where CO_2 exchange is the only contributing factor, and the effect of day-to-day management is minimised. An example of this was in our estimation of the effect a heavy grazing event under wet conditions resulting in soil pugging had on net CO_2 exchange between two grazing events. Another example where the short-term variation of adjacent sites could be used is the quantification of the impact of pasture renewal (provided there was an alignment of pre-treatment management).

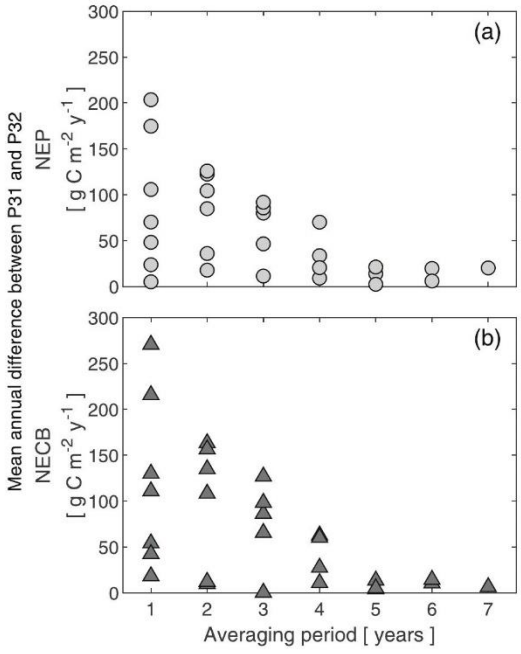

Fig. 7. Illustration of the effect increasing number of measurement years has on reducing the between-paddock variability of (a) NEP and (b) NECB. Each point represents the difference between P31 and P32 for NEP or NECB aggregated across all combinations of annual averaging periods as subsampled from the seven-year dataset. For comparative purposes, data are presented as annual differences, and in absolute terms (i.e. as a positive value irrespective of which paddock had the greater NEP/NECB).

5. Conclusion

In this study, we proposed and tested a method for allowing CO_2 flux data, and by extension NECBs, to be calculated for two adjacent paddocks from a single EC system. This method employed Kormann and Meixner (2001) footprint analysis to assign individual half-hourly flux measurements to one or the other paddock when there was sufficient contribution to the measured flux. Gap-filled flux datasets were then generated for each of the paddocks and expanded to calculate seven years of NECB's for the two paddocks. The seven-year average NECB for P31 and P32 was 1 ± 76 g C m^{-2} y^{-1} and 7 ± 80 g C m^{-2} y^{-1} (mean and 95% confidence interval) respectively, and compared well with the integrated area of both paddocks (NewMix; –14 ± 48 g C m^{-2} y^{-1}). Both paddocks (and NewMix) were considered C neutral and not significantly different from each other over the study period. At the annual timescale, considerable paddock-to-paddock and inter-annual variability were observed, which reduced when averaging over multiple years. Integrating across both paddocks (NewMix) smoothed these management differences, however, inter-annual variability remained large (similar to P31, but smaller than P32).

Use of the methodology employed here to calculate annual NEP/NECB for adjacent paddocks allows a potential avenue for increased replication in EC studies, however, site selection and experimental design require careful consideration. Furthermore, such methodology allows the possibility for two treatments to be compared using a single EC system and thus minimising the effects of spatial variability. However, multiple years of measurements are likely required to determine similarity or difference due to the impact of differing management on individual paddocks. Finally, as rotationally grazed grassland systems are complicated by the influence of day-to-day management decisions, we suggest each paddock should be considered a treatment, and thus advise reporting of cumulative flux data (and/or C balances) at paddock scale rather than aggregating across multiple paddocks.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Supplementary materials

Supplementary material associated with this article can be found, in the online version, at doi:10.1016/j.agrformet.2020.107942.

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Chapter 6:

Summary and conclusions

6.1 Introduction

Agriculture contributes 10-12% of global greenhouse gas (GHG) emissions (Smith *et al.*, 2014) while being the largest contributor to New Zealand's GHG emission profile accounting for 48% of emissions (MfE, 2020). Identifying mechanisms to mitigate GHG emissions has taken on increased importance, with increasing sequestration of atmospheric CO_2 as soil carbon (C) seen as an attractive option (Minasny *et al.*, 2017). Managed agricultural soils, including croplands and grasslands, are considered to have the greatest potential. However, manipulating management aiming to enhance sequestration first requires an understanding of how the various management activities affect soil C stocks.

Within the pasture-based New Zealand dairy industry, supplemental feed is extensively used to account for feed shortfall at times of low pasture growth, and as a year-round supplement to increase total production (Doole, 2014). Furthermore, the use of supplemental feed has significantly increased during the past two decades, including a considerable maize silage contribution (DairyNZ Economics Group, 2016). The use of supplemental feed is one possible mechanism that can increase soil C stocks whereby additional C inputs from excreta attributable to the supplemental feed can be stabilised within the soil (Johnston *et al.*, 2009; Maillard and Angers, 2014). Balancing any gains to soil C through the use of supplement feed is the potential for a loss of soil C during its production. Long-term cropping systems have been shown to have considerably lower soil C stocks than grassland ecosystems (Smith *et al.*, 2016) indicating that production of cropped supplemental feeds such as maize silage likely has a detrimental effect to soil C stocks.

The overarching aim of this thesis was to investigate how supplemental feed altered the C balances of New Zealand dairy farms including aspects of production and use. In this thesis, the methodology used the net ecosystem carbon balance approach (NECB; Chapin *et al.*, 2006), whereby all flows of C into and out of an ecosystem boundary were quantified. The resultant change in ecosystem C was assumed synonymous to a change in soil C when the change in biomass C between the start and end times over which the balance was calculated

was negligible (Paustian *et al.*, 2019). A secondary aim of this thesis was to advance the use of NECB methodology for use in complex, rotationally grazed, agricultural ecosystems primarily through the examination of the scale at which measurements are made, including testing a methodology to calculate NECBs for adjacent paddocks separately.

This chapter provides a summary and conclusions of this thesis. Each of the three objectives are summarised individually (Section 6.2), followed by an overall thesis summary and discussion (Section 6.3) separately covering (i) the overarching theme of how production and use of supplementary feed can affect soil C stocks of New Zealand dairy systems, and (ii) NECB methodology advancements in rotationally grazed pasture systems. Finally, Section 6.4 outlines several aspects of potential future work arising from this thesis.

6.2 Research summary and implications

6.2.1 Objective 1

The first objective of this thesis was:

To quantify the NECB of a dairy farm importing large quantities of supplemental feed (>40% of the cows' diet). This objective tests the hypothesis that large imports of supplemental feed would result in a positive NECB (a presumed gain in soil C). Additionally, the choice of system boundary on the calculation of NECB and subsequent interpretation of the results is examined by calculating the NECB for two system boundaries: (i) an area bound by the footprint of the eddy covariance system ($NECB_{Footprint}$*); and (ii) an area bound by the farm property boundary (*$NECB_{Farm}$*).*

The hypothesis that a large import of supplemental feed would result in a gain in ecosystem C was tested by measuring the NECB for three years at a Waikato dairy farm where between 39 and 50% of the annual animal feed was imported to the farm as palm kernel expeller (PKE) and maize silage. This net import of C in the supplemental feed averaged 526 g C m^{-2} y^{-1}, and the average NECB was 71 ± 77 g C m^{-2} y^{-1} ($NECB_{Farm}$; mean ± uncertainty) which could not be distinguished from zero. All three measurement years returned a positive NECB (indicating a gain in ecosystem C), but a definitive sink of C for the farm only occurred in the first year.

The hypothesised gain in C was based primarily on evidence that additions of farmyard manure (stored and redistributed excreta and effluent) increased soil C stocks (Johnston *et al.*, 2009; Johnston *et al.*, 2017), and that manures have a C retention coefficient of around

12% (Maillard and Angers, 2014), which suggested that an increase in excreta deposition would lead to an increase in ecosystem C stock. Most imported supplementary feed was consumed by the grazing dairy cows, with only a small proportion wasted (5%). Of the ingested supplementary feed, most was lost from the system through the export of product (milk) and to the atmosphere as respired CO_2 and enteric CH_4. The remaining excreted C accounted for ~35% of ingested C which, coupled to the C in the wasted supplementary feed, provided a three-year average net input of 203 g C m^{-2} y^{-1} to the farm system. Using an assumed manure C retention coefficient of 12%, annual sequestration due to imported supplementary feed would have been ~25 g C m^{-2} y^{-1}. Scenario modelling for New Zealand dairy farms by Kirschbaum *et al.* (2017) similarly suggested that small gains in soil C (around 40 g C m^{-2} y^{-1} for this study) could be achieved by importing supplemental feed with the rate dependent on time since the management change to importing supplemental feed. The calculated NECB ($NECB_{Farm}$) for years 2 (35 g C m^{-2} y^{-1}) and 3 (45 g C m^{-2} y^{-1}) were of very similar magnitude to those expected via either calculation method, while year 1 (131 g C m^{-2} y^{-1}) was two to three times higher. Due to the magnitude of the uncertainties relative to the calculated NECB, this study was unable to definitively conclude that importing large quantities of supplemental feed resulted in a gain in ecosystem C. However, the results did confirm that there was no large gain of C despite the import of considerable C embodied in the supplemental feed.

A secondary component of this study was to calculate the NECB for two system boundaries: one surrounding the paddocks within the eddy covariance (EC) flux footprint ($NECB_{Footprint}$), and the other matching the farm boundary ($NECB_{Farm}$). For all years, the calculated NECBs were similar with the $NECB_{Footprint}$ being around 15 g C m^{-2} y^{-1} lower, i.e. $NECB_{Footprint}$ for the three years was 115 ± 85 g C m^{-2} y^{-1}, 19 ± 73 g C m^{-2} y^{-1} and 31 ± 74 g C m^{-2} y^{-1}, while $NECB_{Farm}$ for the three years was: 131 ± 85 g C m^{-2} y^{-1}, 35 ± 73 g C m^{-2} y^{-1} and 45 ± 73 g C m^{-2} y^{-1}). $NECB_{Farm}$ was considered best due to the quality and certainty of the data used in the calculations. While the final NECB was similar regardless of system boundary definition, important components in the calculation, and therefore interpretation, differed. For $NECB_{Farm}$, supplemental feed was a large component of the C balance. However, because most of the supplemental feed was fed outside the EC flux footprint on a dedicated feed pad, it was a minor component of $NECB_{Footprint}$. If interpretation were based on the magnitude of the components used to construct the NECB, supplemental feed would be considered less

important when calculated as $NECB_{Footprint}$ compared to $NECB_{Farm}$, with a complete understanding of the animal C cycle required to identify excreta deposition as the mechanism for supplemental feed C transfer into the footprint. The best choice of system boundary location depends on available data, but careful consideration needs to be given to establishing all flows of C. Moreover, there needs to be a certainty that the measured EC flux is representative of the ecosystem within the entire system boundary. In this study, the EC system was mounted at a height such that the flux footprint incorporated around one-third of the farm. On an annual basis, this was considered sufficiently representative of the farm given the uniform and regimented farm management systems in place. However, the analysis showed clear grazing signals related to the two paddocks immediately adjacent to the EC system suggesting an over-representation of flux derived from these paddocks. In conclusion, similar NECBs can be calculated regardless of the definition of the system boundary, but interpretation may be influenced, and care needs to be given to the representativeness of the measured CO_2 flux.

6.2.2 Objective 2

The second objective of this thesis was:

To calculate the NECB of periodic maize silage production for supplemental feed during the maize cropping period compared to land used for pasture grazing (using data obtained from Objective 3). The hypothesis for this objective is that the periodic cropping nature coupled with conventional tillage and high soil C stocks will result in a large loss of ecosystem C.

The NECBs for the two seasons of maize cropping (referred to as Year 1 and Year 2) measured losses of ecosystem C of –850 g C m^{-2} and –415 g C m^{-2} respectively. In contrast, the NECBs of nearby grazed pasture during the same period was 11 g C m^{-2} and –114 g C m^{-2}. The larger C loss from the maize crop in the first season was due to a smaller NEP (55 g C m^{-2} in Year 1 compared to 240 g C m^{-2} in Year 2), and a more productive crop with a larger export of biomass (–1081 g C m^{-2} in year 1, and –820 g C m^{-2} in Year 2). Poorer growth in year 2 was attributed to soil moisture limitations resulting in reduced biomass production and hence C export. The pasture site was also affected by the dry conditions resulting in the measured net loss of ecosystem C in year 2. Total ecosystem C lost during the two measured maize crops was –1265 g C m^{-2} while for the pasture the C balance was –103 g C m^{-2}. If the pasture site were considered a control, the true effect of the maize cropping was –1162 g C m^{-2}, although as

found in Objective 3 (Section 6.2.3), conclusions drawn using individual paddocks or years from pasture sites should be interpreted cautiously.

The magnitude of ecosystem C loss from the maize crop was similar to that of other studies reporting C balances of maize cropping with full biomass removal (i.e. production for silage and biofuel) (e.g. Béziat *et al.*, 2009; Eichelmann *et al.*, 2016; Poyda *et al.*, 2019). However, this study aimed to quantify the C balance for just the maize crop and exclude the winter period, whereas most previously published studies report annual C balances including winter management. Winter management practices, including winter or cover cropping and leaving fallow, have been demonstrated to increase (Wang *et al.*, 2015), decrease (Jans *et al.*, 2010) or have a negligible effect on NECB (Eichelmann *et al.*, 2016) and, therefore, can obscure the true effect of maize cropping when annual C balances are reported. To quantify the effect that maize silage cropping had, a clear definition of the start- and end-points of the management process was required. While the killing of the previous sward at the beginning of the maize establishment phase was an obvious start point, identifying the definitive end was much more difficult. In this study, the pasture re-establishment following the maize harvest was included as this would not have been required had the maize crop not been planted. A definable and observable endpoint of seedling emergence of the subsequent sward was chosen, with additional C lost between harvest and this point (–49 and –71 g C m^{-2} for the respective years) due to a lack of vegetative cover and thus minimal photosynthetic uptake. Evidence from pasture renewal studies (Rutledge *et al.*, 2017b) indicated C was lost for several more days past seedling emergence and, therefore, losses attributed to the maize silage cropping process may be larger than reported here. Regardless of endpoint, significant quantities of ecosystem C were lost due to maize silage cropping.

This study also further reinforced the role animals play in the C cycle of grazed agricultural systems. Harvest of the maize silage for storage and later use resulted in >90% of the aboveground biomass C being exported, which contrasted with only ~60% being exported (as milk, respired CO_4 and enteric CH_4) from the pasture site. Wastage of pasture through trampling by the grazers coupled with their cycling of the ingested C to deposit excreta resulted in a significant proportion (~40%) of the aboveground biomass C remaining within the paddock confines. Additionally, as the cows consume the harvested and stored maize silage they redistribute the excreted proportion throughout the farm, which equated to an

additional ~20 g C m^{-2} of excreta C being deposited across the entire farm (including the cropping paddocks upon a return to pasture).

The hypothesis for this study that periodic maize silage cropping would result in a large C loss was supported. However, opportunities to reduce these observed losses include minimising the time soil is bare (i.e. without photosynthetic biomass) during both the maize and pasture establishment phases and, although not tested, a reduction in tillage intensity may be beneficial. Furthermore, the periodic nature of the cropping – that is land which was pasture, then cropped for maize silage (in this case for two years), followed by a return to pasture – has the potential for the lost C to be regained under a return to pasture. Existing literature indicates the conversion of long-term continuous cropping systems to pasture results in a gain in C (C recovery; e.g. Conant *et al.*, 2017), and if the same occurs in periodically cropped systems such as this study, longer-term this study site may be C neutral. If the production of maize silage is C neutral in the long-term, the increased biomass production relative to the pasture could lead to C gains from the additional excreta deposited.

6.2.3 Objective 3

The third objective of this thesis was:

To determine the viability of calculating paddock specific CO_2 exchange and NECB of two adjacent paddocks using a single EC system, and thus providing a novel method to test the effect of management practices and/or increase replication.

This study had two purposes: firstly, as a test case for calculating NECB at the paddock scale aimed at minimising the impact of heterogeneous management on primarily NEP, but also NECB. The second purpose was to determine the NECB of a pasture system for comparison with the maize silage production of Objective 2. The study site consisted of two adjacent paddocks (P31 and P32) with an EC system located on the fenceline between them. Flux footprint modelling (Kormann and Meixner, 2001) allowed calculation of the contribution each paddock made to the half-hourly flux, which was then subsequently filtered using a minimum contribution threshold to assign the flux to the dominant source paddock resulting in two datasets. Despite the limited data coverage (average data coverage from the seven years of measurements were 25.9% for P31 and 15.7% for P32), the artificial neural network gap-filling routine performed sufficiently well to allow calculation of annual fluxes for each paddock. Gap-filling of all flux data collected from the integrated area across both paddocks

produced a reference dataset (NewMix; average data coverage of 49.1%), which replicated previously published methods (e.g. Rutledge *et al.*, 2017a, b). Finally, for comparative purposes, a composite flux ($NewMix_{Composite}$) was calculated by multiplying the P31 and P32 annual flux totals by the annual proportion of the flux derived from each respective paddock as obtained from the footprint analysis. Generally, good agreement was found between NewMix and $NewMix_{Compoisite}$ for CO_2, while further validation of the method was provided by the comparison of annual evaporation, which also showed very good agreement despite the regular small differences in the P31 and P32 evaporation totals.

Annual NECB totals for the different years ranged from –150 to 176 g C m^{-2} y^{-1} for P31, –107 to 200 g C m^{-2} y^{-1} for P32 and –110 to 84 g C m^{-2} y^{-1} for NewMix indicating large inter-annual variability. Between-paddock variability was also large which resulted in the smaller range of NewMix annual NECBs. Overall management of the two paddocks was the same, but the timing of management events (particularly grazing events) was a likely contributor to the annual differences between the paddocks. For example, a grazing event in P31 during winter rainfall resulted in pugging damage to the paddock leading to a sustained period of reduced NEP. This management-related influence on the NEP of P31 was not observed in P32 despite being grazed only a couple of days later under similar conditions but with careful management by the farmer. Small differences in the day-to-day management of individual paddocks can influence the calculated NECB of each, however, when integrating across both paddocks these effects are smoothed out as evidenced by the smaller range of NewMix NECB totals. Despite the large inter-annual and between-paddock variability, after seven years of measurements P31, P32 and NewMix datasets had similar average annual NECBs. The average annual NECB of P31 was 1 g C m^{-2} y^{-1}, P32 was 7 g C m^{-2} y^{-1} and NewMix –14 g C m^{-2} y^{-1} and the site was considered C neutral. Analysis of the period over which measurements are made suggested that a minimum of five years was required to overcome between-paddock variability and minimise the size of the difference between the paddocks using this methodology.

This study concluded that calculating paddock specific NECBs from two adjacent paddocks with one EC system is possible. Moreover, due to the influence small day-to-day management differences have on the C balance of each paddock, interpretation is likely improved by calculating paddock specific NECBs. This technique provides opportunities to increase replication (identified as a limitation of NECB studies by Hill *et al.* (2017)), and/or allow for

testing of mitigation strategies with reduced resources. However, limitations of this technique include reduced data coverage (and related increased uncertainty) and a requirement for several years of measurements to detect smaller differences (~50 g C m^{-2} y^{-1}), although larger differences may be determined sooner. Finally, the large inter-annual variability, whether measuring individual paddocks or integrating across both, indicated that conclusions based on a single year of data need to be drawn with extreme caution.

6.3 Thesis summary and conclusions

6.3.1 Supplemental feed production and use

To understand the role supplemental feed plays in the C balance of grazed New Zealand dairy systems, both its production and use need to be considered together rather than separately. In simplest terms, assuming a dairy farm is otherwise C neutral, simply importing supplemental feed from external locations is likely to result in increases in ecosystem C through increased excreta deposition. Regardless of the quantity imported, theoretical annual gains will likely be small (<50 g C m^{-2} y^{-1}), a consensus supported by the first component of this thesis (Wall *et al.*, 2019) and modelling for New Zealand dairy systems by Kirschbaum *et al.* (2017). While beneficial for the farm importing the feed, potential C losses at the site of production should be included (Figure 6.1). For the example of maize silage, the second component of this thesis (Wall *et al.*, 2020a) identified large losses of C associated with its production. However, different forms of supplemental feed that were not studied in this thesis, such as harvested pasture or other arable crops, may have a lesser impact on ecosystem C (e.g. Koncz *et al.*, 2017; Poyda *et al.*, 2019). Finally, the results for maize production presented here are from a periodically cropped system, and therefore may not be transferable to a continuously cropped system. Shepherd *et al.* (2001) found that the rate of C losses may diminish with time in continuously cropped systems and, therefore, the average loss associated with maize silage production from continuously cropped lands may be smaller.

The scenario described above identified the production and use of supplemental feed as separate systems, but the two are commonly used together in New Zealand dairy systems where a cropping phase is included in the pasture renewal process (Densley *et al.*, 2001). Here, individual paddocks with exhausted pastures are first cropped for (e.g.) maize silage, then returned to pasture for several years. In these systems, both the loss of ecosystem C through the production of the crop and gains through the deposition of additional excreta occur within

the same farm system and therefore could be considered together. Consequently, a farm ecosystem C budget for maize silage supplementary feed alone can be constructed. For example, Troughton Farm (Objective 2 study farm) utilised 13 ha of land to grow maize silage annually (equivalent to 6.5% of the farm). Assuming an average annual C loss of –6.3 t C ha^{-1}, the 13 hectares of maize silage grown per year on the farm would result in annual losses of –82.2 t C y^{-1}. The average harvest yield was 9.5 t C ha^{-1} y^{-1} (123.5 t C y^{-1} total) of which approximately one-third (41.2 t C y^{-1}) was returned to the farm ecosystem as excreta after consumption by the grazing animals. The direct net loss of C from the farm (maize production minus excreta deposited) would have been –41 t C y^{-1} or –20 g C m^{-2} y^{-1} averaged across the farm (Figure 6.1). However, assuming only 12% of the excreta C is stabilised (Maillard and Angers, 2014), the remaining 88% (36.3 t C y^{-1}) would be eventually returned to the atmosphere leading to a farm-wide net cost of maize silage use of –77.3 t C y^{-1} (–39 g C m^{-2} y^{-1} averaged across the farm). To summarise, the integrated system of maize silage production and use within the same farm represents a small net loss of C, with the production losses being partially offset by gains to the remainder of the farm via increased excreta deposition following consumption.

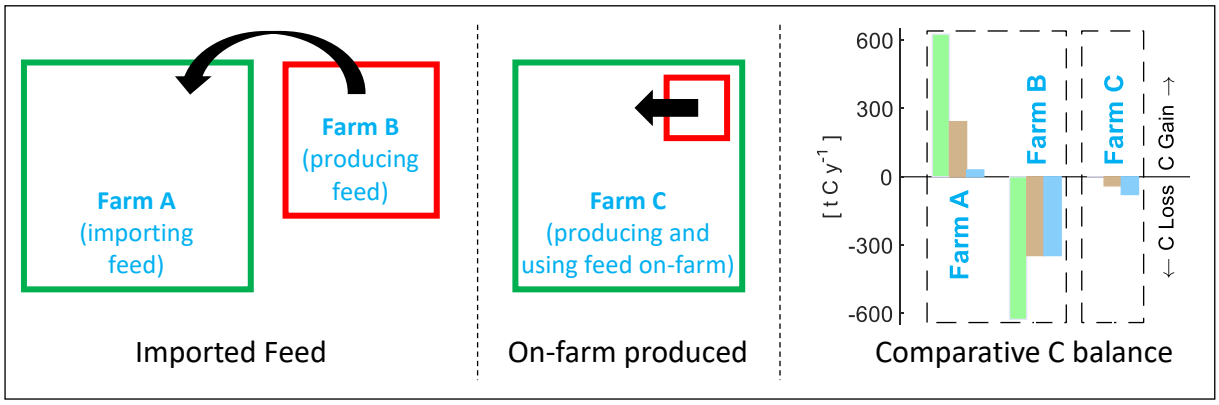

Figure 6.1: Conceptual diagram of the movement of maize silage as supplemental feed and comparative C balances for two farming systems: (1) importing supplemental feed to Farm A that has been produced on Farm B; and (2) on-farm production of supplemental feed via periodic cropping (Farm C). The green squares represent the farm using the supplemental feed, the red squares represent the production area, with the size of the red square to the green square proportional to data obtained from the Ryan Farm (Chapter 3) for imported feed, and Troughton Farm (Chapter 4) for the on-farm produced feed. The black arrows indicate the flow of supplement feed. The comparative C balance represents total C change for the farm and is generated from data in this thesis assuming that continuous cropping of maize silage (as may be likely for Farm B) results in similar C losses to periodic cropping. The green bars are the transfer of feed C, the tan bars the C balance immediately after use (i.e. including that excreted), while the blue bars indicate the estimated long-term C balance following stabilisation of the deposited excreta. Note that any potential recovery of C in the cropland that is returned to pasture is not accounted for.

An advantage of on-farm maize silage production via periodic cropping is that time is available for lost C to be recovered once the maize paddocks are returned to pasture and other areas of the farm are cropped. Typically, reported rates of C gain following a land-use change from croplands to pasture (or grasslands) are around 1 t C ha^{-1} y^{-1} (100 g C m^{-2} y^{-1}; Shepherd *et al.*, 2001; Conant *et al.*, 2017). If it is assumed that C lost during periodic cropping events is recovered at a similar rate and that each maize silage crop loses ~6.3 t C ha^{-1}, only 6-7 years would be required to return to C neutrality, and up to ~15% of the farm could be cropped while allowing sufficient recovery time. Note that, in the instance of the study farm in this thesis (Troughton Farm), paddocks were cropped for maize silage for two years rather than one and, therefore, the expected recovery period could be twice as long. If long-term maize silage production is C neutral, the gains associated with the additional excreta would lead to a net increase in ecosystem C, and therefore have GHG mitigation potential. However, experimentally, both the recovery and long-term gains may be difficult to determine. The third study of this thesis determined annual C balances for seven-years which showed large inter-annual and between-paddock variability. Consequently, short duration (1-3 years) studies would make it difficult to conclusively identify gains whether larger (as in the case of recovery after cropping) or smaller (i.e. from additional excreta deposition). Longer-duration NECB studies may be able to identify recovery of C in the cropped paddocks if accumulation rates approach an average of 1 t C ha^{-1} y^{-1}, however, the smaller gains across the remaining farm areas due to increased excreta deposition may be obscured by the (comparatively) large uncertainties. Moreover, the calculated C balance integrates across management and climate (Ammann *et al.*, 2020), and therefore isolating a single process is difficult.

To summarise, the use of supplemental feed leads to theoretical gains in ecosystem C, which is assumed to translate to increases in soil C stocks. Experimental results showed large quantities of imported supplemental feed did not result in large gains of C, with uncertainties precluding conclusion of small gains. Production of maize silage for supplemental feed resulted in a large loss of C relative to the pasture alternative, but when grown on-farm a small proportion of the loss is offset by additional excreta deposition.

6.3.2 NECB methodology advances in grazed pasture systems

The overarching research focus of this thesis on the role of supplement feed in grazed pasture systems provided an opportunity to advance C balance methodology within these grazed ecosystems. Primarily, the key question was “at which scale could and should system

boundaries be established?" Further to this was the question of whether paddock-scale NECBs could be calculated when an EC system resides on the border between adjacent paddocks allowing for increased replication or paired treatment-control experiments. Within Objective 1, similar NECBs were calculated with system boundaries surrounding both the farm boundary and the paddocks included within the EC flux footprint. In the third study, NECBs were calculated for individual paddocks from a single EC system.

The key conclusion from both studies was that a clear understanding of what the measured CO_2 flux represents is required. When measuring across multiple paddocks, the EC flux is assumed to represent the average flux from all paddocks. While all paddocks contribute to the measured CO_2 flux, this flux is proportional to the distance from the EC system as evidenced by the probability distribution shape of the various flux footprint models (e.g. Kormann and Meixner, 2001; Kljun *et al.*, 2015). Consequently, the measured flux is dominated by the management, and particularly the grazing regime, of those paddocks closest. As in Chapter 3 (Wall *et al.*, 2019), scaling a measured flux that is integrated across multiple paddocks to farm-scale is likely to be more representative of the farm than if the flux was derived from a footprint covering just one or two paddocks. However, the assumption that the flux is representative of the farm does not account for any other management practices occurring outside the flux footprint. The management of the Ryan Farm studied in Chapter 3 was sufficiently uniform and regimented that this assumption was considered valid, but would likely be invalid for many New Zealand farm systems. For example, the Troughton Farm site used 13% of the farm for maize silage production, where the CO_2 flux dynamics were very different to those of the pasture, and therefore, a single measure of CO_2 flux could not be assumed representative of the entire farm. Chapter 5 (Wall *et al.*, 2020b) demonstrated that integrating across two paddocks resulted in an annual flux (both CO_2 and NECB) which was effectively a composite of the individual paddock fluxes. Due to the unequal representation of both paddocks, the composite flux was dominated by the paddock over which the flux footprint was most frequently sourced. Where management is heterogeneous (e.g. asynchronous grazing), the result can be over or under-estimated fluxes on an annual scale, while on the daily scale specific events may be missed. Indeed, Griebel *et al.* (2020) highlight heterogeneity as impacting flux data in general terms, so increasing the homogeneity by limiting management variation by synchronising management across multiple paddocks, or calculating paddock-specific fluxes, would be beneficial. While creating

datasets for individual paddocks significantly reduces available data, the trade-off is increased nuance in the record, and therefore the ability to interpret controlling factors. Additionally, the primary advantage is the ability to capture the specific management of a single paddock while minimising the effect of spatial variability.

Ultimately determining the appropriate scale to measure EC fluxes, and therefore NECB, from grazed pasture systems depends on the purpose of the research. Aside from the consideration of the representativeness of the measured flux, the location of the system boundary dictates both the components of the C balance and the subsequent interpretation. For example, Chapter 3 demonstrated that because most supplement feed was fed on a dedicated feed pad outside the flux footprint, relatively little supplement feed was imported to the flux footprint despite constituting up to half the grazers diet. Consequently, imported supplemental feed was a minor component of the footprint-scale NECB. A full understanding of both the management system and particularly the role the animals play in the cycling of C within the system provided the understanding that they were transferring supplemental feed into the footprint as excreta.

Finally, calculation of paddock-scale fluxes where two adjacent paddocks can be measured with one EC system provides additional opportunities. Aside from reducing management heterogeneity, replication can be increased (Hill *et al.*, 2017), or different treatments could be imposed on each paddock providing a potential tool to test mitigation strategies in agricultural systems. However, a few cautions should be noted. Firstly, considerable variability in the C balance both inter-annually and between-paddock likely requires several years of measurements to provide conclusive results, although larger differences may be detected sooner. Secondly, study sites need to be assessed for suitability for such experiments to ensure regular data are collected from each paddock, and specific climatic conditions are not associated with specific wind sectors.

6.4 Future research

The relationship between supplemental feed production and use in New Zealand dairy farm systems and GHG mitigation is substantially broader than covered in this thesis, with many further avenues that can be explored and questions requiring answers, several of which are outlined in this section.

6.4.1 What is the effect of other forms of supplemental feed on ecosystem C?

In this thesis, maize silage was explored as an example of supplemental feed. While common within the Waikato region where the study farms of this thesis reside, there are many other forms of supplemental feed. These include harvested arable crops and pastures and grazed crops such as fodder beet, turnips, kale and rape (DairyNZ Economics Group, 2016). Improving the knowledge base on how these other forms of supplemental feed affect ecosystem C will allow for improved decision-making to maintain production through supplemental feed production while minimising or mitigating agricultural GHG emissions. Internationally, harvested pasture has been extensively studied (e.g. Senapati *et al.*, 2014; Koncz *et al.*, 2017), and therefore an emphasis on the grazed crops should be the priority. Indeed, fodder beet was the third most common supplementary feed used within the New Zealand dairy industry (DairyNZ Economics Group, 2016), and would be the ideal candidate for further exploration.

6.4.2 Full GHG budgets

The focus of this thesis was on ecosystem C balances, but the impact supplemental feed has on GHGs of N_2O and CH_4 must also be considered. Maize silage has a lower nitrogen content than pasture implying lower animal N intake, and therefore less N excreted and lower N_2O emissions (Dijkstra *et al.*, 2011). Similarly, some studies have indicated enteric CH_4 emissions from consumption of maize silage are lower than for pasture (van Gastelen *et al.*, 2015; Dall-Orsoletta *et al.*, 2019) and, therefore, feeding maize silage may have GHG mitigation potential. Any benefits from feeding maize silage need to be considered in tandem with potential emissions associated with its production. For example, N_2O emissions have been demonstrated to increase with disturbance of the soil during pasture renewal events (Merbold *et al.*, 2014), and similar effects could be expected during maize establishment. Quantification of a full GHG budget for the entire supplemental feed process (production and consumption) is challenging but would provide more certainty on any mitigation potential. Inclusion of all GHGs would be beneficial to not just maize silage, but also studies of any other forms of supplemental feed production.

6.4.3 Life-cycle analysis

Expansion from all GHGs to complete life-cycle analysis would further enhance the understanding of supplemental feed production and use. Harvested supplemental feed involves the use of considerable farm machinery and hence fossil fuel consumption (e.g. Adler

et al., 2007). Moreover, mechanical equipment is used during the establishment, harvest and feeding-out phases while, depending on the source location, transportation may be required for delivery of the feed. For example, considerable supplemental feed imported to the farm studied in Chapter 3 was in the form of palm kernel expeller (PKE), which had been imported to New Zealand. Additionally, fertiliser use can be considerable during crop production (i.e. there were three fertiliser applications per maize crop in the Chapter 4 study) and would need to be included. Accounting for all stages of the supplemental feed process, including those beyond just the direct GHG emissions would provide the clearest picture of the impacts of using supplemental feed to improve the production of animal products within New Zealand systems.

6.4.4 Recovery of ecosystem C following maize silage production

One hypothesis arising from Chapter 4 following calculation of the NECB for maize silage production was that periodic cropping might be long-term C neutral. That is, C lost during the periodic cropping is recovered following a return to permanent pasture. Continuing C balance measurements for several years after a return to pasture would be essential to test this hypothesis. Measurements would need to be made for multiple years to establish long-term trends, rather than simply those occurring due to short-term inter-annual variability including that caused by climatic variations or extremes.

6.4.5 Paddock-specific fluxes for other gases

Chapter 5 demonstrated that EC fluxes of CO_2 and H_2O could be determined from individual paddocks from a single EC system, however, other GHG fluxes such as N_2O and CH_4 were not measured. Expansion of the methods proposed in this thesis could be tested on these other gases at a site with a suitable configuration and ability to measure such trace gases. While gap-filling procedures associated with CO_2 and H_2O are well established (e.g. Moffat *et al.*, 2007) enabling the ability to resolve the low data coverage, such methods may be more challenging for these other gases, particularly N_2O where gap-filling is an emerging area of research (Nemitz *et al.*, 2018; Bigaignon *et al.*, 2020). Ability to calculate paddock-specific fluxes for all GHGs would provide extensive opportunity for testing different agricultural GHG mitigation options.

6.4.6 EC and NECB measurements for mitigation

Eddy covariance and NECB studies have a significant role to play in identifying and testing management-based mitigation opportunities within agricultural systems. As demonstrated in this thesis, these methodologies allow for isolating and quantifying C changes of individual management activity (e.g. maize silage cropping) including providing an improved understanding of the associated flows of C. Effective understanding requires, where possible, limiting the analysis to just the specific management activity in question while avoiding integration across multiple practices, as often occurs when reporting on an annual basis. Calculating paddock-scale fluxes would be more beneficial than integrating across multiple paddocks while, if adjacent paddocks can be measured using a single EC system, there are opportunities including replication, and/or use of treatment-control experiments. However, consideration must be given to the magnitude of any expected difference due to increased uncertainty arising from the reduced data coverage when using the adjacent paddocks approach. Estimation of soil C stock change via soil sampling methods should be included within NECB studies, primarily as supporting evidence (especially where a significant change in C stock is expected) but also for the potential to contribute to a widespread sampling study. NECB studies of agricultural management activities provide complementary information otherwise unobtainable from soil sampling studies, and with careful experimental design should be considered for testing of mitigation options of grazed and arable agricultural ecosystems. Finally, the high temporal data provided by EC and NECB studies is useful for feeding into modelling studies (e.g. Kirschbaum *et al.*, 2017; Liang *et al.*, 2020) to further enhance the understanding of agricultural C cycling, and identify mitigation opportunities.

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Appendix A:

Supplementary materials for Chapter 3

Table S1: Digestibility values used for different feed types. Pasture digestibility values were applied seasonally and estimated based on data from DairyNZ (2017). Digestibility for maize silage, maize husk, and PKE was from analysis of samples taken.

Feed Type	Digestibility
Pasture – Winter	70.3%
Pasture – Spring	75.0%
Pasture – Summer	62.5%
Pasture – Autumn	70.3%
Maize Silage	64.6–70.6%
Maize Husk	66.8%
PKE	70.0%

Table S2: Uncertainty percentages for each parameter used in the calculation of non-CO_2 NECB components. The plausible range was determined as the range of values expected, and the uncertainty applied was half the plausible range and the value used in error propagation calculations. Where a "term" is entered, the uncertainty applied corresponds directly to the matching term in one of the listed equations.

Parameter	Term	Plausible Range	Uncertainty Applied	Notes
Synthetic fertiliser	$F_{Syn\ Fert}$	±50%	±25%	
Leaching	$F_{Leaching}$	±100%	±50%	
Milk	F_{Milk}	±10%	±5%	
Pasture production		±20%	±10%	
Imported supplemental feed mass		±10%	±5%	
C content (measured)		±5%	±2.5%	
C content (estimated)		±10%	±5%	
Digestibility (measured)		±5%	±2.5%	
Digestibility (estimated)		±20%	±10%	
Urine as a percentage of dung		±20%	±10%	
Utilisation (pasture)		±12%	±6%	Derived from uncertainty of ±0.10 (i.e. 0.85 ± 0.10)
Utilisation (feed pad fed supplemental feed)		±5%	±2.5%	Derived from uncertainty of ±0.05 (i.e. 0.95 ± 0.05)
Utilisation (paddock fed supplemental feed)		±19%	±9.5%	Derived from uncertainty of ±0.15 (i.e. 0.80 ± 0.15)
Time spent on paddock/feed pad		±4%	±2%	Plausible range equivalent to 1 hour
Volatile solids component of feed waste	$VS_{Feed\ Waste}$	±50%	±25%	
Volatile solids component of dung	VS_{Dung}	±50%	±25%	
CH_4 Emission factor for feed waste	$EF_{feed\ waste}$	±50%	±25%	
CH_4 Emission factor for dung	EF_{Dung}	±50%	±25%	
CO_2 biogas as a percentage of CH_4 biogas		±50%	±25%	

Table S3: Average animal C balance, and partitioning of on-farm produced and imported supplementary feed for three measurement years. The animal C balance was calculated as the daily total on both a per cow basis and converted to per livestock unit (LU), where the livestock unit was assumed to be 600 kg liveweight. Wastage percentages were calculated using utilisation values of 0.85 for grazed pasture (Macdonald et al., 2008), 0.8 for supplement feed consumed on the grazed paddock, and 0.95 for supplement feed consumed on the dedicated feed pad (DairyNZ, 2017).

				On-farm produced feed		**Imported supplementary feed**	
	C balance per cow [g C cow^{-1} d^{-1}]	**C balance per LU** [g C LU^{-1} d^{-1}]	**Percentage of intake** [%]	**Total** [g C m^{-2} y^{-1}]	**Percentage of intake** [%]	**Total** [g C m^{-2} y^{-1}]	**Percentage of intake** [%]
C Available	-	-	-	680	-	526	-
C Wastage	-	-	-	103	-	27	-
C_{Intake}	6.68	8.01	100%	577	100%	498	100%
C_{Milk}	1.19	1.43	18%	103	18%	89	18%
$C_{Excreta}$	2.32	2.78	35%	197	34%	176	35%
$C_{Respiration}$	2.95	3.54	44%	258	45%	217	44%
$C_{Methane}$	0.22	0.26	3%	19	3%	16	3%

Appendix B:

Supplementary materials for Chapter 4

B.1 Summary of Management Events

B.1.1 P31 Management Events

Table S1: Management events occurring in P31 during the study period. Where applicable, fertiliser name or type and application rate has been specified

Year 1		Year 2	
Date	**Event**	**Date**	**Event**
3-Oct-2016	Grazing	25-Sep-2017	Grazing
24-Oct-2016	Grazing	26-Oct-2017	Grazing
28-Oct-2016	Fertiliser application (custom blend applied at 520 kg ha^{-1})	14 & 15 Nov-2017	Grazing
10-Nov-2016	Grazing	1-Dec-2017	Fertiliser application (liquid N applied at 180 L ha^{-1})
1-Dec-2016	Grazing	9 & 10 Dec-2017	Grazing
22-Dec-2016	Grazing	6-Jan-2018	Grazing
20 to 22-Jan-2017	Grazing	8-Feb-2018	Grazing
21 & 22 Feb-2017	Grazing	14-Mar-2018	Harvest
9-Mar-2017	Fertiliser application (SustaiN and salt applied at 100 kg ha^{-1})		
23-Mar-2017	Grazing		

B.1.2 Maize Crop Management Events

Table S2: Summary of maize silage crop management event dates and application rates (where applicable) from two maize silage cropping seasons on Troughton farm (Waikato, New Zealand). Management events span the period during which carbon balance measurements were made and include key events immediately before and after the cropping season. Nutrient application data are list in Table S 3.

	Year 1		Year 2	
Event	**Date**	**Application Rate**	**Date**	**Application Rate**
Last grazing	24 Aug 2016 (P60)[3] 3 & 4 Sep 2016 (P58)	–	27 & 28 Aug 2016 (P58) [3] 29 & 30 Aug 2016 (P60)	–
Harvest of pasture	19 Sep 2016[3]	–	n/a	–
Herbicide application (killing of existing sward)	23 Sep 2016	–	15 Sep 2017	–
Liquid effluent application	9 to 17 Sep 2016[3]	80 m^3 ha^{-1}	27 Sep to 5 Oct 2017	120 m^3 ha^{-1}
Solid effluent (sludge)/compost application[1]	25, 26 & 27 Sep 2016	31,515 kg ha^{-1}	29 Sep 2017	23,695 kg ha^{-1}
Fertiliser application #1	26 Sep 2016	1335 kg ha^{-1}	29 Sep 2017	737 kg ha^{-1}
Tillage – disk ripper	5 Oct 2016	–	6 Oct 2017	–
Tillage – rototiller	8 & 11 Oct 2016	–	12 & 13 Oct 2017	–
Planting of maize	14 Oct 2016	–	15 Oct 2017	–
Fertiliser application #2 (during planting)	14 Oct 2016	100 kg ha^{-1}	15 Oct 2017	100 kg ha^{-1}
Seedling emergence (maize)	26 Oct 2016	–	26 Oct 2017	–
Herbicide application (weed control)	16 Nov 2016	–	25 Nov 2017	–
Fertiliser application #3	23 Nov 2016	150 kg ha^{-1}	4 Dec 2017	215 kg ha^{-1}
Harvest of maize crop	17 Mar 2017	–	6 Mar 2018	–
Grazing	n/a	–	9 Mar 2018	–
Planting of pasture sward[2]	31 Mar 2017	–	9 Mar 2018	–
Seedling emergence	9 Apr 2017	–	23 Mar 2018	–
Fertiliser application #4	4 May 2017[3]	480 kg ha^{-1}	1 May 2018[3]	183 kg ha^{-1}
First grazing of pasture sward	30 May 2017[3]	–	6 May 2018[3]	–

[1] Compost was mixed with the solid effluent and applied together

[2] Winter crop (annual pasture) in year 1, permanent pasture in year 2

[3] Occurred outside of the period considered in this study

B.2 Maize site nutrient applications

Table S3: Rates of nutrient (C, N, P, and K) application attributable to the maize crop process. Nutrients were applied via liquid effluent, solid effluent (sludge)/compost, and fertiliser. Year 1 includes the crop grown between September 2016 and April 2017, while year 2 was the crop grown between September 2017 and April 2018. Applications indicated with an * are outside the study period. Data have units are in kg nutrient ha^{-1}. Fertiliser applications #1 and #4 were custom blends of several synthetic fertilisers, while application #2 was di ammonium phosphate (DAP), and #3 was sulphate of ammonia (SOA) in year 1, and calcium ammonium nitrate (CAN) in year 2.

	Year 1				Year 2			
Event	**C**	**N**	**P**	**K**	**C**	**N**	**P**	**K**
Liquid effluent application	85*	19*	3*	30*	89	21	4	38
Solid effluent (sludge)/compost application	1642	119	35	109	1525	114	50	118
Fertiliser application #1	132	3	4	59	67	1	1	68
Fertiliser application #2	0	18	20	0	0	18	20	0
Fertiliser application #3	0	28	0	0	5	58	0	0
Fertiliser application #4	48*	9*	0*	1*	5*	0*	0*	1*
Total	***1908***	***195***	***63***	***200***	***1692***	***212***	***75***	***226***
Total (study period only)	**1774**	**167**	**59**	**168**	**1687**	**212**	**75**	**225**

B.3 NECB Maize site component calculation methods

Carbon imported to the maize site as manure (F_{Manure}) included both sludge and compost. The sludge was solid effluent that had been deposited on and accumulated from the use of a dedicated feed pad, while the compost was wasted supplemental feed and sludge that had been collected and composted during the previous year. Determination of the manure C import coupled lab analysis of the dry matter, and carbon (and nutrient) contents with an estimation of the mass of wet material applied to the maize paddocks during the preparation/cultivation period. Manure was evenly spread across the paddocks using specialist machinery in several applications. Due to operational constraints, the mass of all

applications was unable to be measured. Therefore, the mass of several applications was measured using a tractor-mounted scale (Alpha Weighlog 10, RDS Systems Ltd.), and an average application mass determined, which was then assumed representative of all other applications. The total mass of manure (wet) applied was estimated by multiplying the average application mass by the total number of applications.

Liquid effluent ($F_{Effluent}$) was applied to the maize paddocks as part of the effluent management protocol of the farm using a travelling effluent irrigator. In this process, the irrigator made several passes across the maize paddocks over several days (Table S 2). To estimate effluent application depth, a transect of collection containers was firstly placed in travelling irrigator's path. Following irrigation, the volume of each container was measured allowing estimation of the irrigation depth as the average of all sample containers. The collected volume was then analysed for nutrient concentration. Finally, $F_{Effluent}$ was calculated as the product of the number of irrigator runs, the length, and width of each run, the application depth, and C concentration.

$F_{Fertiliser}$ was calculated by multiplying the application rate by the C content. The application rate was reported by the commercial operator who applied the fertiliser, while the C content was either calculated from the chemical composition of the fertiliser (if known), or from lab analysis. Nutrient application data (including C) for events that contribute to F_{Manure}, $F_{Effluent}$, and $F_{Fertiliser}$ are listed in Table S 3.

Biomass removed by grazing ($F_{Grazing,P}$) occurred once in year 2 when one herd of cows grazed the maize paddocks following harvest to remove any pasture growing where maize was not planted (i.e. around the edges of the paddock, etc.) and spilled harvested maize silage. Unlike in a grazed pasture, estimation of the available biomass was difficult, so an alternate approach was used. From grazing and supplement feed records coupled with pasture growth measurements, we estimated the average daily ingested biomass for the herd for the previous five days. We then estimated the biomass removed during the grazing event by multiplying the average daily ingested biomass by the time spent by cows grazing the paddock (in days) and the average C content of the pasture (45%) and maize silage (43.5%).

$F_{Excreta\ returned}$ utilised the same procedure as outlined in Rutledge *et al.* (2017). In brief, the total ingested feed C from the current and previous days was multiplied by the digestibility of the feed (separate values for each feed type as determined by either measurements or book

values (DairyNZ, 2017)) to estimate the mass excreted as dung. An average daily excreta (dung) quantity was then calculated and proportioned to the time spent on the paddock. Total excreta returned was then estimated as dung deposited plus 14% to account for urine deposition (Rutledge *et al.*, 2017).

Harvested maize silage biomass C ($F_{Harvest}$) was estimated by multiplying the dry mass of the harvested biomass multiplied by the C content of the maize silage. Initially, we estimated the total harvested biomass in two ways. Firstly, at twenty random locations immediately before harvest, we measured planting density and destructively sampled one plant from each plot to allow for estimation of total aboveground biomass. The process was repeated immediately post-harvest to estimate the non-harvested residual allowing for the determination of total harvested biomass. The second approach was to measure all biomass as it was fed to the grazing animals utilising the tractor-mounted scales. Both methods proved problematic with the sampling approach yielding an unrealistically high harvested biomass quantity while measuring all harvested material as it was fed out was unrealistically low. The same pattern occurred in both years. Further confirmation was provided by the comparison of the ratio of total aboveground biomass to GPP with the first method suggesting almost all GPP was required for aboveground biomass production, and the second method too little. Furthermore, communication with the seed company indicated that the sampling method can produce wildly inconsistent results while weighing all maize silage as fed out excludes a 10-20% loss that can occur during the ensiling process (I. Williams pers comm, 2019). Consequently, we chose to estimate harvested biomass as the total biomass weighed using the second method (tractor scales) plus correction factors. The correction factors were an additional 15% to account for losses during the ensiling process, and a wastage component for year 1 of 7% as estimated by the farmer due to decomposed and thus not used biomass resulting from water damage to the stored maize silage. The C content of the maize silage was measured by laboratory analysis to be 43.5% in year 1 and 43.1% in year 2, while multiple samples were collected to determine the dry matter content. Total harvested biomass was determined as the estimated (wet) multiplied by the dry matter and carbon contents.

$F_{Leaching}$ was calculated by multiplying the annual drainage below 0.6 m soil depth by the dissolved organic carbon concentration. Annual drainage was determined using the Woodward drainage model (Woodward *et al.*, 2001) utilising measured evaporation and rainfall from the EC site. Dissolved organic carbon was assumed to have a concentration of

3.7 μg C L^{-1} as determined by Sparling *et al.* (2016) from samples taken on areas of the farm which included the pasture site, but not including the maize site (the Waihou soil of the maize site was represented within the sampling areas, however). It should be noted that while not measured, the rooting depth of the maize crop likely extended below 0.6 m. We expect the effect of this on the $F_{leaching}$ term to be small, however, we have included a (relatively) large uncertainty to account for this difference.

B.4 References

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Appendix C:

Supplementary materials for Chapter 5

C.1 Nutrient Application

Table S1: Nutrient applications for experimental paddocks. Fertiliser is commercially supplied synthetic fertiliser; manure is duck, chicken and goat manure (largely composed of animal bedding (wood chip and straw) and excrement); compost is waste supplementary feed and collected solid effluent collected and composted; sludge is solid effluent collected from the feed pad. All data are in units of kg ha^{-1} yr^{-1}.

Year	Type	Paddock 31			Paddock 32		
		N	P	K	N	P	K
2012	Fertiliser	31	0	0	32	0	0
	Manure	74	37	39	78	39	43
	TOTAL	*105*	*37*	*39*	*110*	*39*	*43*
2013	Fertiliser	33	20	38	33	20	38
	Manure	16	8	14	16	8	14
	TOTAL	*49*	*28*	*52*	*49*	*28*	*51*
2014	Fertiliser	1	2	1	1	2	1
	Manure	97	34	130	60	21	80
	TOTAL	*98*	*36*	*131*	*61*	*23*	*81*
2015	Fertiliser	27	0	0	26	0	0
	Compost	6	2	4	5	2	3
	TOTAL	*33*	*2*	*4*	*31*	*2*	*3*
2016	Fertiliser	52	2	27	52	2	27
	TOTAL	*52*	*2*	*27*	*52*	*2*	*27*
2017	Fertiliser	37	1	2	37	1	2
	TOTAL	*37*	*1*	*2*	*37*	*1*	*2*
2018	Fertiliser	9	3	18	9	3	18
	Sludge	9	2	12	0	0	0
	Compost	3	1	1	3	1	1
	TOTAL	*21*	*6*	*31*	*12*	*4*	*19*
Average	TOTAL	56	16	41	50	14	32

C.2 Data Coverage

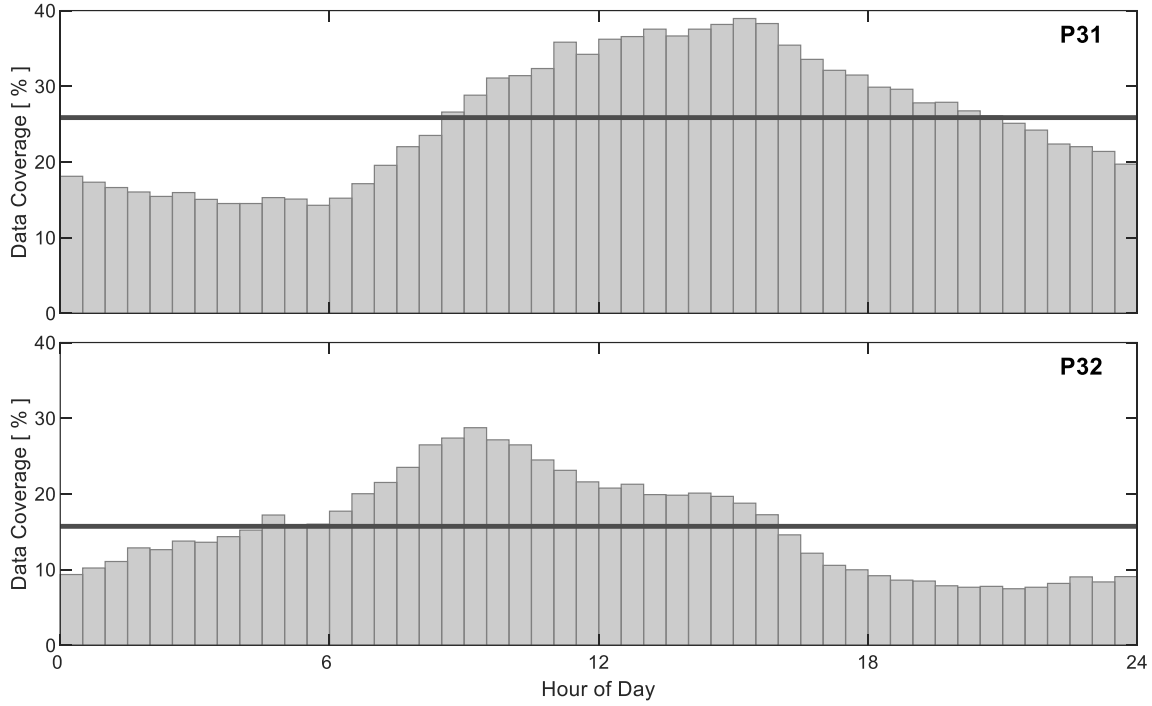

Figure S1: Average data coverage binned by half-hour of the day for P31 (top) and P32 (bottom) for 2012-2018. The dark grey horizontal line represents the mean data coverage of the entire dataset.

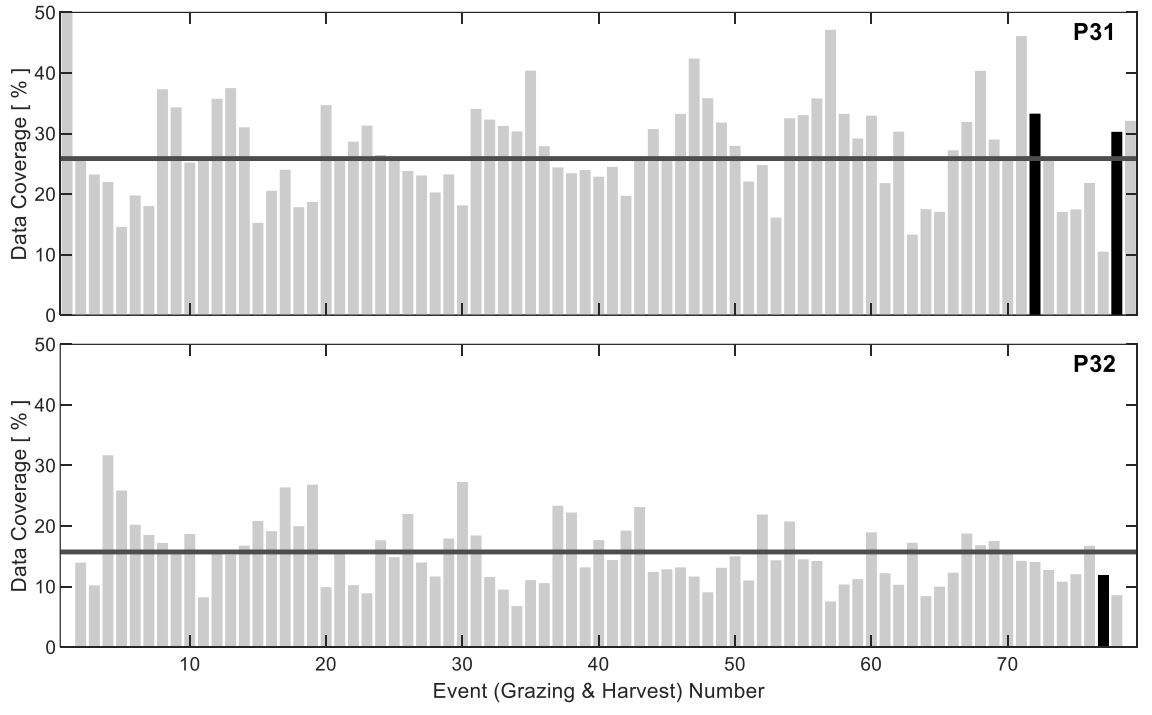

Figure S2: Data coverage for the time interval between all grazing and harvest management events for P31 (top) and P32 (bottom). Grey bars represent grazing events, while the black bars represent harvest events. The dark grey horizontal line represents the mean data coverage of the entirety of each dataset. No data coverage was available for event number 1 for P32 as this (grazing) event occurred on 1 Jan 2012 (i.e. the first day of measurement).

Table S2: Data coverage for P31, P32 and NewMix datasets for a range of conditions and seasons for 2012-2018. Rainfall half-hours were those with recorded rainfall (>=0.2 mm). Sunny half-hours were considered those with an incoming shortwave solar radiation of greater than 0.67 * calculated solar irradiance. Cloudy half hours were those which were classified as neither rainfall or sunny. Seasons were defined as: summer – 1 Dec to 28 (or 29) Feb; autumn – 1 Mar to 31 May; winter – 1 Jun to 31 Aug; spring – 1 Sep to 30 Nov.

Condition/season	P31	P32	NewMix	Number of half-hours in data (sub)set
All data	25.9%	15.7%	49.1%	122,736
Day-time	32.4%	20.1%	64.6%	60,574
Night-time	19.5%	11.4%	34.0%	62,162
Rainfall	28.8%	9.1%	42.0%	10,718
Cloudy[1]	34.6%	17.3%	64.4%	38,989
Sunny[1]	23.9%	25.0%	59.2%	16,835
Summer	31.5%	12.5%	53.5%	30,336
Autumn	22.5%	17.0%	44.0%	30,912
Winter	19.3%	18.1%	41.3%	30,912
Spring	30.3%	15.2%	56.8%	30,576

[1] Day-time only

C.3 ΔNEE Gap-filling Example

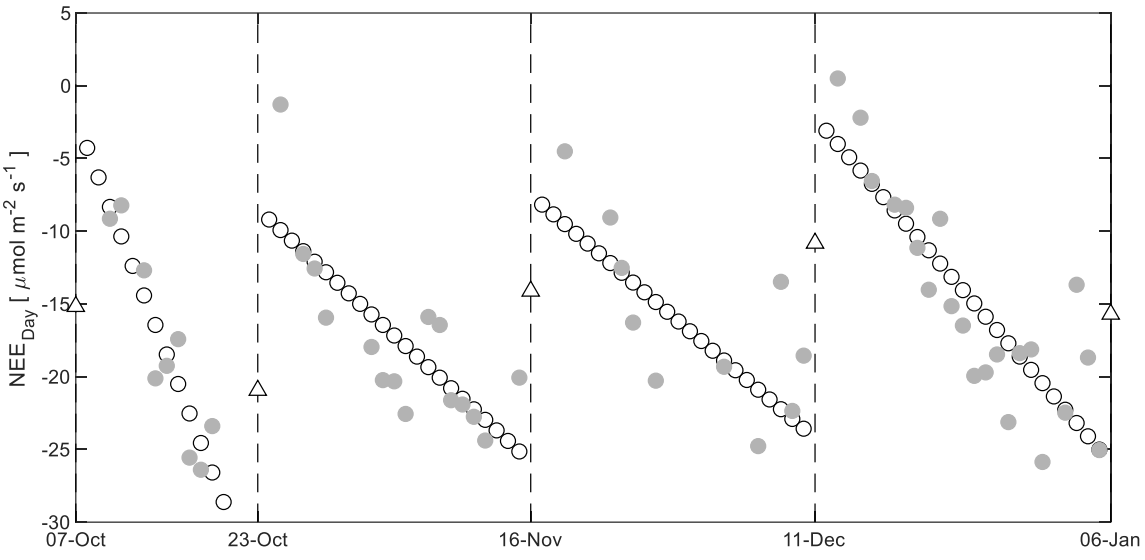

Figure S3: Mean daytime NEE during high-light conditions for P31 illustrating the effect of grazing. Measured daytime NEE are represented with filled circles. Gaps in daytime NEE between grazing events were filled using linear regression (calculated separately for each inter-grazing period; open circles), and linear interpolation (open triangles) during the grazing period. Dates of grazing are the dashed vertical lines.

Appendix D:

Co-authorship forms

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Co-Authorship Form

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Chapter 3 has been published in the journal *Agriculture, Ecosystems and Environment*. The title of this paper is: Carbon budget of an intensively grazed temperate grassland with large quantities of imported supplemental feed

Nature of contribution by PhD candidate	Collaboration on project conception; data collection, filtering and gap filling; conducted data analysis, interpretation and writing of the manuscript
Extent of contribution by PhD candidate (%)	90%

CO-AUTHORS

Name	Nature of Contribution
David Campbell	Collaboration on project conception, data analysis and interpretation; provided comments on manuscript versions
Paul Mudge	Collaboration on data analysis and interpretation; provided comments on manuscript versions
Susanna Rutledge	Collaboration on project conception, data analysis and interpretation
Louis Schipper	Collaboration on project conception, data analysis and interpretation; provided substantial comments on manuscript versions

Certification by Co-Authors

The undersigned hereby certify that:

- the above statement correctly reflects the nature and extent of the PhD candidate's contribution to this work, and the nature of the contribution of each of the co-authors; and

Name	Signature	Date
David Campbell	Dave Campbell	18/08/2020
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Chapter 4 has been published in the journal *Agriculture, Ecosystems and Environment*. The title of this paper is: Quantifying carbon losses from periodic maize silage cropping of permanent temperate pastures

Nature of contribution by PhD candidate	Collaboration on project conception; data collection, filtering and gap filling; conducted data analysis, interpretation and writing of the manuscript
Extent of contribution by PhD candidate (%)	90%

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Chris Morcom	Collaboration on project conception, data collection
Paul Mudge	Collaboration on project conception, data analysis and interpretation; provided comments on manuscript versions
Louis Schipper	Collaboration on project conception, data analysis and interpretation; provided substantial comments on manuscript versions

Certification by Co-Authors

The undersigned hereby certify that:

- the above statement correctly reflects the nature and extent of the PhD candidate's contribution to this work, and the nature of the contribution of each of the co-authors; and

Name	Signature	Date
David Campbell	Dave Campbell	18/08/2020
Chris Morcom		18/08/2020
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Please indicate the chapter/section/pages of this thesis that are extracted from a co-authored work and give the title and publication details or details of submission of the co-authored work. Chapter 5 has been published in the journal *Agricultural and Forest Meteorology*. The title of this paper is: Temperate grazed grassland carbon balances for two adjacent paddocks determined separately from one eddy covariance system	
Nature of contribution by PhD candidate	Collaboration on project conception; data collection, filtering and gap filling; conducted data analysis, interpretation and writing of the manuscript
Extent of contribution by PhD candidate (%)	90%

CO-AUTHORS

Name	Nature of Contribution
David Campbell	Collaboration on project conception, data analysis and interpretation; provided comments on manuscript versions
Paul Mudge	Collaboration on project conception, data analysis and interpretation; provided comments on manuscript versions
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Certification by Co-Authors

The undersigned hereby certify that:

- the above statement correctly reflects the nature and extent of the PhD candidate's contribution to this work, and the nature of the contribution of each of the co-authors; and

Name	Signature	Date
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Paul Mudge	P L Mudge	18/08/2020
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